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Conserving Biodiversity on Native Rangelands: Symposium Proceedings

August 17, 1995

Fort Robinson State Park, Nebraska

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Abstract: These proceedings are the result of a symposium, "Conserving biodiversity on native rangelands" held on August 17, 1995 in Fort Robinson State Park, NE. The purpose of this symposium was to provide a forum to discuss how elements of rangeland biodiversity are being conserved today. We asked, "How resilient and sustainable are rangeland systems to the increasing demands of a growing human population and to extended periods of drought?" Key programs and issues, identified by a program committee, were addressed by researchers and managers. Their papers provide research results, management findings, and describe management programs currently used to conserve rangeland biodiversity.

Keywords: biodiversity, rangeland, sustainability, drought, conservation

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Fort Robinson State Park, Nebraska

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Contents

Introduction

Rangelands embody biological diversity of profound ecological and social significance, yet it is the biological diversity of forests and wetlands that has been the focus of research by scientists and concern by the public. Recently, a broad array of people, from ecologists and biologists to ranchers and recreationists, have begun to realize the importance of rangeland conservation and biological diversity. Although these groups may not always share a common vision of rangelands, they share a common interest in the land that will foster a better understanding and appreciation of the value of diverse and healthy rangelands.

Ranchers have long practiced conservation of rangeland biological diversity. Most recognize the importance of both warm and cool season grasses to round out their forage programs, and many have noticed that in some years one grass will do poorly while another will thrive, thus balancing the production. Ranchers depend on native grasses coming back on their own after drought or a bad grasshopper year; some species will return quicker than others. Looking toward the future, ranchers manage their grass for a diverse rangeland community, not a monotypic one. This is conservation of rangeland biological diversity at the grass roots level.

Together, scientists and rangeland managers are traveling to new levels of conservation of rangeland biodiversity, but the journey has some formidable challenges. Herbivory, fire, drought, and other natural events and processes historically shaped rangeland biodiversity and ecological processes long before human action. However, human influence on the range has complicated and interrupted many naturally occurring mechanisms. The use and control of fire has altered its frequency and intensity. The pattern, frequency, and intensity of herbivory by large animals has been modified by the conversion from free-ranging bison and other large ungulates to confined domestic livestock and a proliferation of livestock water developments. Cultivation has fragmented and isolated rangelands and often natural processes no longer function. An insidious challenge to rangeland biodiversity is the invasion of exotic plants into native range often at the expense of native biota.

The purpose of this symposium was to provide a forum to discuss how elements of rangeland biodiversity are being conserved today. We asked, "How resilient and sustainable are rangeland systems to the increasing demands of a growing human population and to extended periods of drought?" One way to begin answering this question is to look at our successes and failures in conserving all parts of rangeland systems. Key programs and issues, identified by a program committee, were addressed by researchers and managers. Their papers, which have received statistical and peer review, are presented here and provide research results, management findings, and describe management programs currently used to conserve rangeland biodiversity. The paper "Gap Analysis in the Great Plains: A Large-Scale Geographic Strategy for Conservation of Biodiversity" by Dennis Jelinski, Michael Jennings, and James Merchant was withdrawn by the authors before publication of this workshop proceedings.

This symposium was held concurrently with the Annual Meeting of the Central Mountains and Plains Section of The Wildlife Society. We thank the organizers of that event for suggesting this symposium. Thanks are also extended for the well-attended field trip to review northern swift fox management in southwestern South Dakota that concluded the workshop.

A Neotropical Migratory Bird Prioritization for National Forests and Grasslands

Dick Roth[1] and Richard Peterson[2]

Abstract.—The Rocky Mountain Region of the USDA Forest Service provides nesting habitat for 146 species of neotropical migratory birds. Interactive, prioritization databases were developed for each National Forest and National Grassland in the Region to assist land managers in making informed decisions about resource allocations. The data was processed using Paradox software. This paper summarizes the uses and application of the database for the Oglala and Ft. Pierre National Grasslands.

METHODS

We used data provided by Colorado Bird Observatory and ranked according to the Partners-In-Flight (PIF) ranking scheme for initial prioritization of neotropical migratory birds (NTMBs). The approach ranks species by their relative susceptibility to extinction (Carter and Barker 1993, Hunter et al. 1993). There are many factors that contribute to extinction probability. The PIF prioritization scheme uses seven criteria as the most important in gauging a species susceptibility to extirpation or extinction: 1) importance of area of consideration (IA), (percentage of a species range that is within a state or geographic area under consideration); 2) global abundance (GA); 3) the degree of threat to the species' persistence on the breeding ground (TB); 4) the degree of threat to species' persistence on the wintering ground (TW); 5) breeding distribution (BD); 6) extent of wintering distribution (WD); 7) population trend in area of consideration (PT); based upon U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) data. Each of the seven criteria is weighted equally. An individual species is assigned a score in each of the seven categories ranging from one (low concern) to five (high concern). Each species is ranked according to the average of the seven scores. The importance of area score (IA) was modified for our use to include a rank based upon the percentage of the area under consideration which meets breeding habitat requirements for a given species.

Uncertainty values are assigned to each species in conjunction with values assigned for threats to breeding (TBU) and wintering (TWU), and population trend (PTU). These uncertainty values reflect the extent of the available information for each of the associated criteria. They indicate the extent and location of gaps in our knowledge of neotropical migrant biology. These values help us differentiate between species with definite management concerns and those requiring additional monitoring or research in order to more clearly reflect their status.

Several criteria were modified for the Oglala and Ft. Pierre National Grasslands. Population trend (PT) and Population trend uncertainty scores were determined from USFWS Breeding Bird Survey (BBS) for the 10-year and 26-year scores. Data from physiographic region 39 (Missouri Plateau-Unglaciated) were used for both grasslands. Other population trend data more specific to the area under consideration can be used for these criteria if available. Threats to breeding habitat (TB) and Threats to breeding habitat uncertainty (TBU) criteria provided by PIF were used (Carter and Barker 1993). Additionally, known local threats were also considered such as reduction of prairie dog towns as a threat to burrowing owl habitat. In this case, a TB score of 5 was used because loss of prairie dog towns would result in elimination of burrowing owl habitat (Peterson 1994).

Several methods have been developed to determine priorities for community based conservation (Millsap et al. 1990, Master 1991, Reed 1992). The technique developed by Partners in Flight is essentially one that ranks individual species first, and secondarily ranks habitats based on individual species scores grouped by habitat preference. This ranking can then be used to develop and justify community based conservation programs. The determina-

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tion of breeding occurrence and habitat preference of neotropical migratory landbirds on the Oglala and Ft. Pierre Grasslands was made using local expertise.

The habitat types and conditions developed for the Grasslands and assigned to each species have three levels:

1) Appropriate habitat contains six major breeding bird habitat types. They include trees/woodlands, shrubs/shrublands, grass/grasslands, edge-tree/grass-shrub/grass, wetlands and special topographic structure.
2) Suitable habitat, in general, additional conditions are needed for appropriate habitat to be suitable breeding habitat for a given species. For grasslands, additional conditions could be related to a given height and density of grasses or forbs. For trees/woodland habitat, additional conditions could include deciduous trees, cavities or a multi-layered canopy.
3) Special conditions includes topographic structures such as cliffs and cutbanks, but also includes features such as riparian areas and prairie dog towns.

These habitat categories enable development of habitat ranking based on a species' use of a wide variety of habitat types and variables.

Coding used for habitats and special features is as follows:

Habitats T-(t)rees/woodlands, coni(f)erous, (d)eciduous, (o)ld growth, m(u)ltilayer canopy, and
(c)avities.
E-(e)dge, tree-grass/shrub-grass.
S-(s)hrubs, (b)ig sagebrush, (2) thorny shrubs-esp. plum,
G-(g)rass/grasslands-open areas-esp. s(h)ort and/or sparse, t(a)ll and/or de(n)se, mi(x)ed/mid.
W-(w)etlands/(w)ater-(1)riparian, (m)arsh/tall emergent, (3) wet meadow-tall grass/short emergent.

Specials s(P)ecial-topo/structure-(4)cliffs/caves/ledges and cutbanks, (5)buildings/bridges/chimneys and bird houses, (6)islands/bare shores.
s(p)ecial-other-(7)prairie dog towns, (i.e. burrows/bare ground/short grass and associated prey), forest fire locations-(B)urned areas, esp. large with tall snags, (9) cropland-esp alfalfa, (0)old crow/magpie nests.

The mix of numbers and letters used in the coding may appear to be confusing; however, familiarization with the application of those codes as displayed in the habitat columns of the accompanying tables reveals that they provide a logical fit.

RESULTS

The Oglala and Ft. Pierre National Grasslands support 79 and 68 species of neotropical migratory landbirds which regularly nest there, or a combined total of 84 regular nesters. These are listed in Appendix 1 and 2 along with all associated prioritization scores for the seven criteria and some of the associated uncertainty scores. Species with R10 or R26 ranks of 3.00 or greater should be given high priority for management considerations (Thompson et al. 1993). Analysis of the data reveals that 18 of the 84 species have a R10 or R26 rank of 3.00 or greater (Appendix 1 and 2). The R10 and R26rank scores along with importance of area, threats to breeding and breeding distribution scores help to provide a framework for setting management priorities. As an illustration, the chestnut-collard longspur has high R10 and R26 rank scores but has an importance of area (IA) score of only 3.00.

Consequently, other species with higher IA scores should be given higher management priority. The two top ranked species on both grasslands (burrowing owl and ferruginous hawk) have a preference for short-grass prairie and prairie dog towns. Other species on these two grassland have a preference for tall and mixed-grass prairie. Consequently, management of the National Grassland units for a diversity of heights and would provide habitat for both species.

The database contains scores for each criterion, for each species, for each unit where they are likely to occur. It is important that the data for each unit be analyzed separately for more specific insights into the top priority species and habitat for each unit. For

example, what is the importance of the habitat on the unit being analyzed for a given species. What are the threats to that habitat? What is the status and trend of that habitat?

This prioritization system reveals that the highest ranked habitat on the Oglala National Grassland is big sagebrush and that is based on one species (table 1). The next highest ranked habitat is short and mixed-grass prairie and prairie dog towns respectively. These habitats support six and four high priority (= > 3) species respectively. Edge habitat and riparian habitat are both important because of the diversity of species that they support. These values are based upon the relative susceptibility to extinction of species found in each habitat. Information on species as presented in table 2 should also be considered along with the habitat information when weighing the consequences of management actions.

A total of 12 species from the Oglala National Grassland have a R26 Rank of 3.00 or greater. Brewer's Sparrow is the species in big sagebrush habitat which causes the high habitat rank in table 1. The rank of 1 for importance of area score (IA) indicates that only a small portion of the Oglala National Grassland provides suitable breeding habitat for Brewer's Sparrows. The two top-ranked species use prairie dog towns and the top five species also short to mixed grass prairie habitats.Therefore, the highest priority habitats for NTMBs on the Oglala National Grassland should be those that support these species.

Table 1. Habitat association scores for the Oglala National Grassland based on R26 species ranks.

Habitat	$\leq$3	<3 to 2	<1.99	# Species	Average score	Total score
Short/Mix Grass	6	1	1	8	3.08	24.71
Prairie Dog Towns	4	2	1	7	2.94	20.57
Mix/tall Grass	2	7		9	2.81	29.00
Trees Deciduous	2	8	2	12	2.48	29.71
Shrub Big Sage	1			1	3.14	3.14
Shrub Dense		5		5	2.60	13.00
Edge	1	15	7	23	2.32	53.41
Water/marsh		7	4	11	2.18	24.00
Riparian	2	15	5	22	2.36	51.99

Table 2. Species on the Oglala National Grassland with R10 or R26 =$\geq$3.00.

Species	Hab	IA	AB	TB	BD	R10	R26
Burrowing Owl	Gh7	5	4	5	3	3.57	3.86
Long-billed Curlew	Gxh7	5	3	3	4	3.86	3.71
Chestnut-collared Longspur	Gxh	3	3	3	4	3.29	3.57
Lark Bunting	Gxhs	5	2	3	4	3.29	3.43
Ferruginous Hawk	Gxht7	3	4	4	3	3.29	3.29
Black-billed Cuckoo	Tds12	2	3	4	3	3.29	3.14
Bobolink*	Ga39	1	2	4	3	3.14	3.14
Brewer's Sparrow	Sb	1	2	4	3	3.00	3.14
Loggerhead Shrike	Es2	3	3	4	2	3.00	3.14
Dickcissel*	Ga9	1	2	4	3	2.86	3.00
Great Crested Flycatcher	Tdc1	1	2	4	3	3.00	3.00
Prairie Falcon	Gxh47	4	3	3	3	3.14	3.00

** Species found in the area but not confirmed nester on National Grassland.*

Similar analysis of the data for the Ft Pierre National Grassland reveals somewhat different results (table 3). Ft Pierre is in a higher precipitation area and has taller grasses and more deciduous trees than the Oglala National Grassland. Bird species diversity is greater across habitat types than on the Oglala National Grassland and mixed/tall grass habitat higher priority. The burrowing owl is the highest ranked species on both units (table 4). Dickcissel , bobolink, grasshopper sparrow, northern harrier and upland sandpiper had higher prioritization scores on the Ft. Pierre National Grassland. Management of prairie dog towns and short grass habitat should have some priority on Ft. Pierre, but management for mixed to tall grass habitat is of higher priority based on this analysis.

Table 3. Habitat association scores for the Ft. Pierre National Grassland based on R26 species ranks.

Habitat	≥3	>3 to 2	>1.99	# Species	Average score	Total score
Short/Mix Grass	5	1	1	7	3.06	21.43
Prairie Dog Towns	3	2	1	6	2.81	16.86
Mix/Tall Grass	5	5		10	3.13	31.29
Trees Deciduous	2	9	2	13	2.50	32.58
Shrub Dense	1	5		6	2.26	13.57
Edge	1	9	7	17	1.98	33.70
Water/marsh	1	8	5	14	2.01	28.13
Riparian	3	14	5	22	2.39	52.58

Table 4. Species on Ft. Pierre National Grassland with R10 or R26 scores ≥ 3.00.

Species	Hab	IA	TB	BD	AB	R10	R26
Burrowing Owl	Gh7	4	5	3	5	3.57	3.86
Baird's Sparrow* (Historic)	Gx3	4	5	5	0	3.86	3.71
Chestnut-collared Longspur	Gxh	3	3	4	3	3.29	3.57
Dickcissel	Ga9	2	3	3	5	3.29	3.43
Ferruginous Hawk	Gxht7	4	4	3	4	3.43	3.43
Lark Bunting	Gxhs	2	3	4	5	3.29	3.43
Bobolink	Ga39	2	3	3	3	3.29	3.29
Long-billed Curlew*	Gxh7	3	4	4	1	3.43	3.29
Bell's Vireo*	Sn12	3	4	3	1	3.14	3.14
Black-billed Cuckoo	Tds12	3	4	3	2	3.29	3.14
Grasshopper Sparrow	Gxa	2	2	2	5	2.57	3.00
Great Chrested Flycatcher*	Tdc1	2	4	3	1	3.00	3.00
Loggerhead Shrike	Es2	3	4	2	2	2.86	3.00
Northern Harrier	Gasm	3	3	1	5	3.00	3.00
Sprague's Pipit* (historic)	Gxa	3	5	4	0	3.00	3.00
Upland Sandpiper	Gx	3	2	3	5	3.14	3.00

** Species found in the area but not confirmed nester on National Grassland.*

CONCLUSIONS

The PIF species ranking system is a helpful tool in establishing priorities for Neotropical Migratory Bird species and habitat based management efforts for those species. It should not replace human judgment or additional information which might be important in setting resource priorities. Refinement of the PIF data as was done on the Oglala and Ft. Pierre National Grasslands with local expertise increases the utility value of the system. Only a few analysis examples were given here. However, an endless variety of queries can be used to tease additional information from the data.

ACKNOWLEDGMENTS

We acknowledge Greg Schenbeck, Wildlife Management Biologist for the Nebraska National Forest for providing information and suggestions to make this report possible.

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Appendix 1. Prioritization scores for the Neotropical Migratory Landbirds of the Oglala National Grasslands.

Species	Hab	AB	TB	TBU	TW	BD	IA	PT26	PTU26	PT10	PTU10	R10	R26
American Goldfinch	Tdes1	1.00	2.00	3.00	1.00	1.00	2.00	3.00	3.00	2.00	3.00	1.43	1.57
American Kestrel	Ec8	1.00	1.00	2.00	2.00	1.00	4.00	1.00	1.00	2.00	3.00	1.71	1.57
American Robin	Ethw	1.00	1.00	1.00	1.00	1.00	2.00	2.00	2.00	4.00	3.00	1.57	1.29
Barn Swallow	Pgw5	1.00	1.00	1.00	2.00	1.00	2.00	1.00	1.00	5.00	1.00	1.86	1.29
Belted Kingfisher	W4	2.00	4.00	2.00	2.00	1.00	1.00	4.00	3.00	3.00	4.00	2.00	2.14
Black-billed Cuckoo	Tds12	3.00	4.00	3.00	3.00	3.00	2.00	3.00	3.00	4.00	3.00	3.29	3.14
Black-headed Grosbeak	Tds1	2.00	3.00	4.00	2.00	3.00	2.00	2.00	3.00	2.00	2.00	2.57	2.57
Blue Grosbeak	Sn2	3.00	3.00	3.00	2.00	2.00	1.00	4.00	3.00	3.00	4.00	2.43	2.57
Bobolink	Ga39	2.00	4.00	2.00	3.00	3.00	1.00	5.00	2.00	5.00	2.00	3.14	3.14
Brewer's Blackbird	Es29	2.00	3.00	5.00	2.00	3.00	1.00	3.00	3.00	3.00	3.00	2.29	2.29
Brewer's Sparrow	Sb	2.00	4.00	3.00	4.00	3.00	1.00	5.00	1.00	4.00	3.00	3.00	3.14
Brown-headed Cowbird	Egsm	1.00	1.00	1.00	1.00	1.00	5.00	1.00	1.00	1.00	1.00	1.71	1.71
Burrowing Owl	Gh7	4.00	5.00	2.00	3.00	3.00	5.00	4.00	3.00	2.00	3.00	3.57	3.86
Cedar Waxwing	Ts	2.00	2.00	3.00	2.00	2.00	1.00	4.00	3.00	3.00	4.00	2.00	2.14
Chestnut-collared Longspur	Gxh	3.00	3.00	4.00	4.00	4.00	3.00	4.00	3.00	2.00	3.00	3.29	3.57
Chipping Sparrow	Efs	1.00	3.00	4.00	2.00	1.00	3.00	4.00	3.00	4.00	3.00	2.29	2.29
Cliff Swallow	Pw45	2.00	2.00	4.00	2.00	1.00	1.00	3.00	3.00	3.00	3.00	2.00	2.00
Common Nighthawk	Eh	2.00	2.00	4.00	2.00	1.00	5.00	3.00	3.00	4.00	3.00	2.43	2.29
Common Poorwill	Ef4	3.00	2.00	4.00	3.00	3.00	2.00	3.00	4.00	3.00	4.00	2.71	2.71
Common Yellowthroat	Wms1	1.00	3.00	2.00	2.00	1.00	2.00	4.00	3.00	5.00	2.00	2.29	2.14
Cooper's Hawk	To1	3.00	3.00	3.00	3.00	1.00	1.00	3.00	4.00	3.00	5.00	2.29	2.29
Dickcissel	Ga9	2.00	4.00	3.00	2.00	3.00	1.00	5.00	1.00	4.00	3.00	2.86	3.00
Eastern Bluebird	Ec85	2.00	3.00	2.00	3.00	3.00	1.00	3.00	4.00	3.00	4.00	2.43	2.43
Eastern Kingbird	E	1.00	1.00	2.00	3.00	2.00	3.00	2.00	3.00	1.00	1.00	2.00	2.14
Eastern Phoebe	Td15	2.00	4.00	4.00	3.00	3.00	1.00	3.00	4.00	3.00	5.00	2.57	2.57
Ferruginous Hawk	Gxht7	4.00	4.00	3.00	3.00	3.00	3.00	3.00	3.00	3.00	3.00	3.29	3.29
Golden Eagle	Et47	3.00	2.00	2.00	2.00	2.00	5.00	4.00	3.00	2.00	3.00	2.57	2.86
Grasshopper Sparrow	Gxa	2.00	2.00	3.00	2.00	2.00	4.00	5.00	1.00	2.00	3.00	2.43	2.86
Gray Catbird	Sn12	2.00	4.00	2.00	2.00	2.00	2.00	4.00	2.00	2.00	3.00	2.43	2.71
Great Crested Flycatcher	Tdc1	2.00	4.00	3.00	4.00	3.00	1.00	3.00	4.00	3.00	4.00	3.00	3.00
Horned Lark	Gh7	1.00	1.00	1.00	1.00	1.00	5.00	2.00	2.00	2.00	3.00	1.71	1.71
House Wren	Tc15	1.00	2.00	1.00	1.00	1.00	2.00	1.00	1.00	1.00	2.00	1.29	1.29
Indigo Bunting	Tds1	1.00	3.00	3.00	2.00	3.00	1.00	3.00	4.00	3.00	4.00	2.29	2.29
Killdeer	Gh67	1.00	2.00	2.00	1.00	1.00	5.00	4.00	1.00	5.00	1.00	2.29	2.14
Lark Bunting	Gxhs	2.00	3.00	3.00	3.00	4.00	5.00	4.00	2.00	3.00	3.00	3.29	3.43
Lark Sparrow	E	3.00	2.00	3.00	2.00	2.00	3.00	3.00	3.00	1.00	2.00	2.29	2.57
Lazuli Bunting	Ts1	2.00	3.00	4.00	2.00	3.00	2.00	4.00	3.00	2.00	3.00	2.57	2.86
Loggerhead Shrike	Es2	3.00	4.00	4.00	4.00	2.00	3.00	3.00	3.00	2.00	2.00	3.00	3.14
Long-billed Curlew	Gxh7	3.00	3.00	4.00	3.00	4.00	5.00	4.00	3.00	5.00	1.00	3.86	3.71
Long-eared Owl	Efo0	3.00	3.00	3.00	3.00	1.00	1.00	3.00	5.00	3.00	5.00	2.14	2.14

Appendix 1 (Continued).

Species	Hab	AB	TB	TBU	TW	BD	IA	PT26	PTU26	PT10	PTU10	R10	R26
Marsh Wren	Wm	2.00	4.00	2.00	4.00	3.00	2.00	3.00	4.00	3.00	4.00	2.71	2.71
Merlin	Ef0	4.00	4.00	4.00	3.00	2.00	2.00	3.00	4.00	3.00	4.00	2.86	2.86
Mountain Bluebird	Ec85	2.00	3.00	3.00	3.00	3.00	2.00	4.00	2.00	3.00	3.00	2.57	2.71
Mourning Dove	Ew	1.00	1.00	1.00	1.00	1.00	4.00	1.00	1.00	3.00	3.00	1.71	1.43
N. Rough-winged Swallow	Pw4	3.00	3.00	3.00	2.00	1.00	2.00	4.00	3.00	3.00	3.00	2.43	2.57
Northern Flicker	Ec8	1.00	2.00	1.00	1.00	1.00	2.00	4.00	2.00	3.00	3.00	1.57	1.71
Northern Harrier	Gasm	3.00	4.00	3.00	4.00	1.00	3.00	4.00	3.00	4.00	3.00	2.86	2.86
Northern Mockingbird	Eds12	1.00	2.00	2.00	1.00	2.00	1.00	3.00	4.00	3.00	4.00	1.71	1.71
Northern Oriole	Tds1	2.00	3.00	3.00	2.00	2.00	2.00	3.00	3.00	3.00	3.00	2.43	2.43
Orchard Oriole	Tds1	3.00	3.00	3.00	2.00	3.00	2.00	2.00	2.00	4.00	3.00	2.86	2.57
Ovenbird	Tu	2.00	4.00	4.00	4.00	3.00	1.00	3.00	4.00	3.00	4.00	2.86	2.86
Pine Siskin	Tfe	1.00	2.00	3.00	1.00	2.00	2.00	3.00	4.00	3.00	4.00	1.71	1.71
Prairie Falcon	Gxh47	3.00	3.00	3.00	3.00	3.00	4.00	3.00	3.00	4.00	3.00	3.14	3.00
Red-eyed Vireo	Tdu1	1.00	4.00	4.00	2.00	2.00	1.00	3.00	3.00	3.00	4.00	2.29	2.29
Red-tailed Hawk	Etg	1.00	2.00	2.00	2.00	1.00	5.00	3.00	3.00	3.00	3.00	2.14	2.14
Red-winged Blackbird	Wms1	1.00	2.00	1.00	1.00	1.00	2.00	4.00	2.00	4.00	3.00	1.71	1.71
Rock Wren	P4	3.00	2.00	2.00	2.00	3.00	3.00	4.00	2.00	4.00	3.00	2.71	2.71
Rufous-sided Towhee	Sn	1.00	3.00	4.00	2.00	2.00	2.00	4.00	3.00	4.00	3.00	2.29	2.29
Say's Phoebe	G45	3.00	2.00	3.00	3.00	2.00	3.00	3.00	3.00	3.00	3.00	2.71	2.71
Sharp-shinned Hawk	Tfo	3.00	3.00	2.00	3.00	1.00	1.00	3.00	4.00	3.00	4.00	2.14	2.14
Short-eared Owl	Gasm	3.00	4.00	4.00	4.00	1.00	2.00	2.00	3.00	4.00	3.00	2.71	2.43
Swainson's Hawk	Gxt9	3.00	2.00	2.00	3.00	2.00	5.00	2.00	1.00	2.00	3.00	2.86	2.86
Tree Swallow	Ec15	2.00	4.00	3.00	3.00	1.00	1.00	2.00	3.00	3.00	3.00	2.29	2.14
Turkey Vulture	E4	1.00	2.00	4.00	2.00	1.00	4.00	2.00	2.00	4.00	3.00	2.14	1.86
Upland Sandpiper	Gx	3.00	2.00	3.00	3.00	3.00	4.00	2.00	3.00	3.00	3.00	3.00	2.86
Vesper Sparrow	Gxs	3.00	3.00	4.00	2.00	2.00	2.00	4.00	3.00	4.00	3.00	2.57	2.57
Violet-green Swallow	Efc4	2.00	3.00	3.00	2.00	3.00	2.00	3.00	4.00	3.00	4.00	2.57	2.57
Warbling Vireo	Td1	2.00	3.00	4.00	2.00	2.00	2.00	5.00	1.00	4.00	3.00	2.57	2.71
Western Kingbird	E	1.00	1.00	2.00	2.00	3.00	3.00	1.00	1.00	1.00	2.00	2.14	2.14
Western Meadowlark	Gx7	1.00	2.00	2.00	3.00	2.00	5.00	3.00	3.00	3.00	3.00	2.43	2.43
Western Tanager	Tf	2.00	3.00	4.00	2.00	3.00	2.00	3.00	4.00	3.00	4.00	2.57	2.57
Western Wood-Pewee	T	2.00	3.00	4.00	3.00	2.00	2.00	2.00	2.00	1.00	1.00	2.43	2.57
White-throated Swift	P4	3.00	2.00	3.00	2.00	3.00	1.00	3.00	4.00	3.00	4.00	2.43	2.43
Willow Flycatcher	Sn12	3.00	4.00	3.00	3.00	3.00	1.00	3.00	5.00	4.00	3.00	3.00	2.86
Yellow Warbler	Tds1	1.00	4.00	3.00	2.00	1.00	2.00	2.00	3.00	2.00	3.00	1.86	1.86
Yellow-billed Cuckoo	Tds12	3.00	4.00	3.00	3.00	2.00	2.00	2.00	3.00	3.00	4.00	2.86	2.71
Yellow-breasted Chat	Sn12	2.00	3.00	3.00	3.00	2.00	2.00	3.00	3.00	1.00	1.00	2.29	2.57
Yellow-headed Blackbird	Wm	3.00	4.00	2.00	3.00	3.00	2.00	2.00	3.00	4.00	3.00	3.14	2.86
Yellow-rumped Warbler	Tf	1.00	2.00	2.00	1.00	1.00	2.00		4.00	3.00	4.00	1.71	1.29

Appendix 2. Prioritization scores for the Neotropical Migratory Landbirds of the Ft. Pierre National Grasslands.

Species	Hab	AB	TB	TBU	TW	BD	IA	PT26	PTU26	PT10	PTU10	R10	R26
American Goldfinch	Tdes1	1.00	2.00	3.00	1.00	1.00	2.00	3.00	3.00	2.00	3.00	1.43	1.57
American Kestrel	Ec8	1.00	2.00	2.00	2.00	1.00	4.00	1.00	1.00	2.00	3.00	1.86	1.71
American Robin	Ethw	1.00	1.00	1.00	1.00	1.00	2.00	2.00	2.00	4.00	3.00	1.57	1.29
Baird's Sparrow(historic)	Gx3	4.00	5.00	4.00	4.00	5.00	0.00	4.00	3.00	5.00	2.00	3.86	3.71
Bank Swallow	Pw4	3.00	3.00	4.00	2.00	1.00	1.00	2.00	3.00	2.00	2.00	2.14	2.14
Barn Swallow	Pgw5	1.00	1.00	1.00	2.00	1.00	2.00	1.00	1.00	5.00	1.00	1.86	1.29
Bell's Vireo	Sn12	3.00	4.00	3.00	4.00	3.00	1.00	3.00	4.00	3.00	4.00	3.14	3.14
Belted Kingfisher	W4	2.00	4.00	2.00	2.00	1.00	1.00	4.00	3.00	3.00	4.00	2.00	2.14
Black-billed Cuckoo	Tds12	3.00	4.00	3.00	3.00	3.00	2.00	3.00	3.00	4.00	3.00	3.29	3.14
Black-headed Grosbeak	Tds1	2.00	3.00	4.00	2.00	3.00	1.00	2.00	3.00	2.00	2.00	2.43	2.43
Blue Grosbeak	Sn2	3.00	3.00	3.00	2.00	2.00	1.00	4.00	3.00	3.00	4.00	2.43	2.57
Bobolink	Ga39	2.00	3.00	2.00	3.00	3.00	3.00	5.00	2.00	5.00	2.00	3.29	3.29
Brown-headed Cowbird	Egsm	1.00	1.00	1.00	1.00	1.00	5.00	1.00	1.00	1.00	1.00	1.71	1.71
Burrowing Owl	Gh7	4.00	5.00	2.00	3.00	3.00	5.00	4.00	3.00	2.00	3.00	3.57	3.86
Cedar Waxwing	Ts	2.00	3.00	3.00	2.00	2.00	1.00	4.00	3.00	3.00	4.00	2.14	2.29
Chestnut-collared Longspur	Gxh	3.00	3.00	4.00	4.00	4.00	3.00	4.00	3.00	2.00	3.00	3.29	3.57
Chipping Sparrow	Efs	1.00	3.00	4.00	2.00	1.00	1.00	4.00	3.00	4.00	3.00	2.00	2.00
Cliff Swallow	Pw45	2.00	2.00	4.00	2.00	1.00	1.00	3.00	3.00	3.00	3.00	2.00	2.00
Common Nighthawk	Eh	2.00	2.00	4.00	2.00	1.00	5.00	3.00	3.00	4.00	3.00	2.43	2.29
Common Yellowthroat	Wms1	1.00	3.00	2.00	2.00	1.00	2.00	4.00	3.00	5.00	2.00	2.29	2.14
Dickcissel	Ga9	2.00	3.00	3.00	2.00	3.00	5.00	5.00	1.00	4.00	3.00	3.29	3.43
Eastern Bluebird	Ec85	2.00	3.00	2.00	3.00	3.00	1.00	3.00	4.00	3.00	4.00	2.43	2.43
Eastern Kingbird	E	1.00	1.00	2.00	3.00	2.00	3.00	2.00	3.00	1.00	1.00	2.00	2.14
Eastern Phoebe	Td15	2.00	4.00	4.00	3.00	3.00	1.00	3.00	4.00	3.00	5.00	2.57	2.57
Ferruginous Hawk	Gxht7	4.00	4.00	3.00	3.00	3.00	4.00	3.00	3.00	3.00	3.00	3.43	3.43
Grasshopper Sparrow	Gxa	2.00	2.00	3.00	2.00	2.00	5.00	5.00	1.00	2.00	3.00	2.57	3.00
Gray Catbird	Sn12	2.00	4.00	2.00	2.00	2.00	1.00	4.00	2.00	2.00	3.00	2.29	2.57
Great Crested Flycatcher	Tdc1	2.00	4.00	3.00	4.00	3.00	1.00	3.00	4.00	3.00	4.00	3.00	3.00
Horned Lark	Gh7	1.00	1.00	1.00	1.00	1.00	5.00	2.00	2.00	2.00	3.00	1.71	1.71
House Wren	Tc15	1.00	2.00	1.00	1.00	1.00	2.00	1.00	1.00	1.00	2.00	1.29	1.29
Indigo Bunting	Tds1	1.00	3.00	3.00	2.00	3.00	1.00	3.00	4.00	3.00	4.00	2.29	2.29
Killdeer	Gh67	1.00	2.00	2.00	1.00	1.00	5.00	4.00	1.00	5.00	1.00	2.29	2.14
Lark Bunting	Gxhs	2.00	3.00	3.00	3.00	4.00	5.00	4.00	2.00	3.00	3.00	3.29	3.43
Lark Sparrow	E	3.00	2.00	3.00	2.00	2.00	2.00	3.00	3.00	1.00	2.00	2.14	2.43
Least Flycatcher	Td1	3.00	4.00	5.00	4.00	2.00	1.00	3.00	3.00	2.00	3.00	2.71	2.86

Appendix 2 (Continued).

Species	Hab	AB	TB	TBU	TW	BD	IA	PT26	PTU26	PT10	PTU10	R10	R26
Loggerhead Shrike	Es2	3.00	4.00	4.00	4.00	2.00	2.00	3.00	3.00	2.00	2.00	2.86	3.00
Long-billed Curlew	Gxh7	3.00	4.00	4.00	3.00	4.00	1.00	4.00	3.00	5.00	1.00	3.43	3.29
Long-eared Owl	Efo0	3.00	3.00	3.00	3.00	1.00	1.00	3.00	5.00	3.00	5.00	2.14	2.14
Marsh Wren	Wm	2.00	4.00	2.00	4.00	3.00	1.00	3.00	3.00	3.00	4.00	2.57	2.57
Mourning Dove	Ew	1.00	1.00	1.00	1.00	1.00	4.00	1.00	1.00	3.00	3.00	1.71	1.43
N. Rough-winged Swallow	Pw4	3.00	3.00	3.00	2.00	1.00	2.00	4.00	3.00	3.00	3.00	2.43	2.57
Northern Flicker	Ec8	1.00	2.00	1.00	1.00	1.00	2.00	4.00	2.00	3.00	3.00	1.57	1.71
Northern Harrier	Gasm	3.00	3.00	3.00	4.00	1.00	5.00	4.00	3.00	4.00	3.00	3.00	3.00
Northern Mockingbird	Eds12	1.00	2.00	2.00	1.00	2.00	1.00	3.00	4.00	3.00	4.00	1.71	1.71
Northern Oriole	Tds1	2.00	3.00	3.00	2.00	2.00	2.00	3.00	3.00	3.00	3.00	2.43	2.43
Orchard Oriole	Tds1	3.00	3.00	3.00	2.00	3.00	3.00	2.00	2.00	4.00	3.00	3.00	2.71
Red-eyed Vireo	Tdu1	1.00	4.00	4.00	2.00	2.00	1.00	3.00	3.00	3.00	4.00	2.29	2.29
Red-tailed Hawk	Etg	1.00	2.00	2.00	2.00	1.00	5.00	3.00	3.00	3.00	3.00	2.14	2.14
Red-winged Blackbird	Wms1	1.00	2.00	1.00	1.00	1.00	3.00	4.00	2.00	4.00	3.00	1.86	1.86
Rock Wren	P4	3.00	3.00	2.00	2.00	3.00	1.00	4.00	2.00	4.00	3.00	2.57	2.57
Rufous-sided Towhee	Sn	1.00	3.00	4.00	2.00	2.00	2.00	4.00	3.00	4.00	3.00	2.29	2.29
Savannah Sparrow	Gx3	3.00	4.00	4.00	3.00	1.00	1.00	5.00	1.00	5.00	1.00	2.71	2.71
Say's Phoebe	G45	3.00	2.00	3.00	3.00	2.00	1.00	3.00	3.00	3.00	3.00	2.43	2.43
Short-eared Owl	Gasm	3.00	3.00	4.00	4.00	1.00	5.00	2.00	3.00	4.00	3.00	3.00	2.71
Sprague's Pipit(historic)	Gxa	3.00	5.00	5.00	3.00	4.00	0.00	3.00	3.00	3.00	3.00	3.00	3.00
Swainson's Hawk	Gxt9	3.00	2.00	2.00	3.00	2.00	5.00	2.00	1.00	2.00	3.00	2.86	2.86
Tree Swallow	Ec15	2.00	3.00	3.00	3.00	1.00	1.00	2.00	3.00	3.00	3.00	2.14	2.00
Turkey Vulture (no nest?)	E4	1.00	2.00	4.00	2.00	1.00	1.00	2.00	2.00	4.00	3.00	1.71	1.43
Upland Sandpiper	Gx	3.00	2.00	3.00	3.00	3.00	5.00	2.00	3.00	3.00	3.00	3.14	3.00
Vesper Sparrow	Gxs	3.00	3.00	4.00	2.00	2.00	1.00	4.00	3.00	4.00	3.00	2.43	2.43
Warbling Vireo	Td1	2.00	4.00	4.00	2.00	2.00	2.00	5.00	1.00	4.00	3.00	2.71	2.86
Western Kingbird	E	1.00	1.00	2.00	2.00	3.00	3.00	1.00	1.00	1.00	2.00	2.14	2.14
Western Meadowlark	Gx7	1.00	2.00	2.00	3.00	2.00	5.00	3.00	3.00	3.00	3.00	2.43	2.43
Willow Flycatcher	Sn12	3.00	4.00	3.00	3.00	3.00	1.00	3.00	5.00	4.00	3.00	3.00	2.86
Yellow Warbler	Tds1	1.00	4.00	3.00	2.00	1.00	2.00	2.00	3.00	2.00	3.00	1.86	1.86
Yellow-billed Cuckoo	Tds12	3.00	4.00	3.00	3.00	2.00	1.00	2.00	3.00	3.00	4.00	2.71	2.43
Yellow-breasted Chat	Sn12	2.00	3.00	3.00	3.00	2.00	1.00	3.00	3.00	1.00	1.00	2.14	2.43
Yellow-headed Blackbird	Wm	3.00	3.00	2.00	3.00	3.00	2.00	2.00	3.00	4.00	3.00	3.00	2.71

Greater Prairie Chicken Nesting Habitat, Sheyenne National Grassland, North Dakota

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Abstract.—Greater prairie chicken (*Tympanuchus cupido pinnatus*) populations and habitats have declined dramatically in the Great Plains. The Sheyenne National Grassland (SNG) has the largest population of greater prairie chickens in North Dakota, but this population has declined over the past 15 years. Lack of nesting habitat has been identified as a significant factor contributing to the decline in greater prairie chicken populations throughout their range. We used the Habitat Suitability Index (HSI) model for greater prairie chickens to evaluate the nesting habitat conditions on the SNG. This population of greater prairie chickens appears to sustain itself on the brink of extirpation by nesting in the few areas that provide nesting cover and in private alfalfa fields. Encroachment of woody plants into the SNG, changes in private land-use patterns, removal of forage by domestic livestock contribute to the low suitability of the SNG for nesting by greater prairie chickens.

INTRODUCTION

The Sheyenne National Grassland (SNG) is approximately 28,745 ha of federally administered prairie in southeastern North Dakota. Within its administrative boundary there are an additional 25, 910 ha of interspersed private cropland and prairie. The SNG contains the largest population of greater prairie chickens (*Tympanuchus cupido pinnatus*) in the state of North Dakota (Kobriger et al. 1987). Greater prairie chickens are not native to the SNG, but are considered a naturalized immigrant in North Dakota (Johnson and Knue 1989). Prairie chickens apparently moved into North Dakota from the north-central part of the United States during the Euro-American settlement in the 1870's and 1880's (Johnson and Knue 1989, Evans 1968). Greater prairie chicken populations and their habitats (native tall grass prairie) have declined to a small fraction of their historical range (Hjertaas et al. 1993, Samson and Knopf 1994). Thus, the population of greater prairie chickens on the SNG has both regional and national importance.

Numbers of prairie chickens on the SNG increased from the early 1960's through the early 1980's (Kobriger et al. 1987). Since then, prairie chicken numbers on the SNG have declined from a high of 410 males in 1983 to a low of 84 males in 1994 (Kobriger et al. 1987, unpubl. data, Sheyenne National Grassland, Lisbon, ND). State and federal natural resource management agencies, and conservation groups are concerned that management of the SNG may be contributing to the decline in the greater prairie chicken population. Lack of suitable nesting habitat has been identified as the most significant factor limiting populations of greater prairie chickens across their range (Kirsch 1974, Westemeir 1973) and in North Dakota (Svedarsky 1979).

Habitat suitability index (HSI) models are an accepted method for quantifying species' habitats as numerical index (Schamberger et al. 1982). Biological and habitat information are synthesized to formulate index values between zero (unsuitable) and one (optimum) for habitat requisites considered important to a species (U.S. Fish and Wildlife Service 1980). We conducted HSI analyses to assess habitat conditions for greater prairie chickens on the SNG at three scales: 1) the western portion of the SNG and adjacent private lands, 2) the Durler/Venlo Management unit, and 3) areas $\leq$1.6 km of the 14 active booming grounds.

METHODS

The HSI model for greater prairie chickens (Prose 1985) identifies two habitat components, nesting cover and winter food, as the most important habitat com-

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ponents for prairie chickens. The HSI for nesting cover is based on grassland vegetation height/density (expressed as visual obstruction measurements on a pole, Robel et al. 1970) for nesting cover in the spring (figure 1).

We mapped the lowland, midland, and upland grassland vegetation types (Manske and Barker 1987) on 1:24,000 aerial photos of the SNG. Most nesting by greater prairie chickens on the SNG occurs within 1.6 km of leks (Newell et al. 1987). The Custer National Forest Land Management Plan (U.S. Forest Service, Custer National Forest, Billings, MT, 1986) requires that nesting habitat for prairie grouse be assessed within 1.6 km of leks. During October and November, 1994, we estimated height/density of vegetation in these vegetation types from 81 transects within 1.6 km of greater prairie chicken leks in the northern and western portion of the SNG. At each of 10 stations on each transect, we recorded the height that vegetation obstructed 100 percent of a pole (VOR) marked in 0.5 dm increments when viewed from four directions (at 90° azimuths) at a distance of 4 m and a height of 1 m from the pole (Robel et al. 1970). VORs were averaged for each station and the average among stations was used to estimate transect VORs. We placed six transects in upland vegetation, 51 transects in midland vegetation and 26 transects in lowland vegetation. Data from these transects were used as VOR estimates in the mapped vegetation polygons they were collected in. For all other mapped vegetation

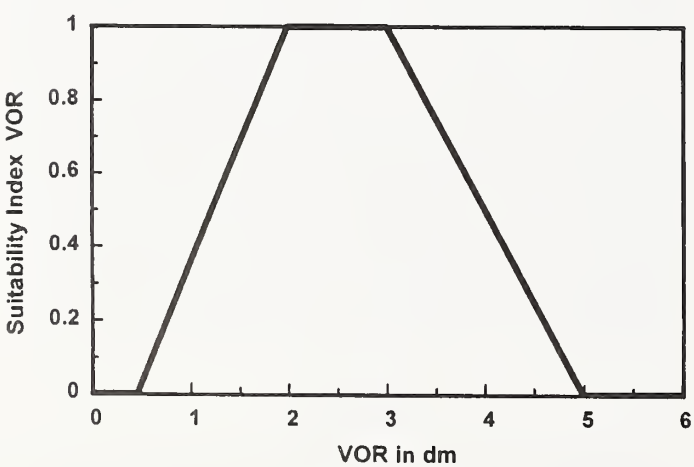

Figure 1. Relationship between average 100 percent obstruction of pole (VOR) marked in 0.5 dm increments and next cover suitability index for greater prairie chickens (from Prose 1985).

polygons, these VOR data served as calibrations for ocular estimates of five VOR classes (0 - 0.50 dm, 0.51 - 1.0 dm, 1.01 - 1.5 dm, 1.51 - 2.0 dm, and >2.0 dm) during field reconnaissance. Maps of vegetation and VOR class assignments were transferred to 1:24,000 U.S. Geological Survey maps and the area of each vegetation was planimetered for use in the HSI estimates.

HSI for nesting cover is estimated in three steps (Prose 1985). First, a suitability index is estimated from the midpoint of the VOR classes of each vegetation type i (SI_{VORi}). Second, the percent of area providing equivalent optimal nesting habitat (EONH) is calculated using:

$$EONH = \sum_{i=1}^{n} (SI_{VOR_i})(N_i)$$

where n = total number of vegetation types, and N_i = percent of the area in vegetation type i. Third, HSI for nest cover is calculated from:

$$HSI = \frac{(0.735 * EONH) - 21.4}{37}$$

Characteristics of vegetation and winter snow accumulation influence the structure of vegetation in the spring for nesting by greater prairie chickens. VOR measurement collected in the fall decrease prior to spring nesting. This decrease is proportional to the height of vegetation and for the range of VOR 0.5 - 2.0 dm varies from 7-40 percent in mixed grass prairie (G. Schenbeck pers. commun., Nebraska National Forest, Chadron, NE). Over winter VOR losses on the SNG are probably different, but data are lacking. We selected 15 percent over-winter VOR losses to estimate spring nesting cover based on fall VOR estimates because the VORs for the SNG are near the lower end of the range.

Western SNG Analysis

The western part of the SNG includes most of the prairie chicken leks. This area included 3433 ha of private land and 8984 ha of SNG administered lands. We calculated the HSI for this analysis unit to show estimated contributions to the HSI for prairie chickens from adjacent private lands. VOR class informa-

tion was available for only 5738 ha (64 percent) of the SNG lands in this analysis unit. We assumed the mapped VOR classes were representative of the remaining of the western SNG and used these data for HSI calculations in this analysis unit. For private lands in the western SNG analysis unit we assumed: 1) CRP land had VOR class >2.0 dm; 2) hay and alfalfa had VOR cover classes <0.5 dm because of mowing approximately the third week of June that destroys existing nests and most young hatched birds; and 3) grazed pasture had VOR cover class 0.51-1.0 dm.

Durler/Venlo Management Unit

The Durler/Venlo management unit includes 3645 ha in nine range management allotments in the western SNG. The Durler/Venlo unit is a subset of the prairie chicken range in the western portion of the SNG. It includes the larger leks, highest prairie chicken numbers, and the greatest number of prairie chicken leks not shared by sharp-tailed grouse (*Tympanuchus phasianellus jamesi*). Most of the Durler/Venlo management unit is $\leq$1.6 km from a prairie chicken lek. This portion of SNG has complete vegetation classification and mapping.

We excluded vegetation communities that were not available for nesting by greater prairie chickens from the HSI for the Durler/Venlo management unit. This HSI analysis presents a complete picture of the nesting habitat for this area. We assigned vegetation types to mapped polygons using the dominant vegetation community in the polygons. Within these polygons, vegetation communities not capable of producing 1.5 dm VOR measurements or that are usually flooded (Manske and Barker 1987, Newell et al. 1987) were considered unavailable for nesting by greater prairie chickens. The area in each polygon assigned to a VOR class did not include unsuitable areas. For example, lowland vegetation communities dominated by species such as *Carex lanulosa* were considered unavailable because in most years the ground is flooded. Upland vegetation communities dominated by species such as *Boutelou gracilis* were considered unavailable for prairie chicken nesting because they are not capable of producing at least 1.5 dm VOR in most years.

Area Surrounding 14 Active Leks

The area within 1.6 km of active leks includes most of the nesting habitat of greater prairie chickens. This scale of analysis allowed us to evaluate HSI for areas of known greater prairie chicken occurrences. This level of analysis included the area surrounding active greater prairie chicken leks and we expected HSI from this analysis should equal or exceed the HSI's from the blocks of SNG that included areas >1.6 km from leks and unused areas.

RESULTS

Western Sheyenne National Grassland

The 12,445 ha in the western SNG had 24 percent EONH (table 1), less than the minimum considered necessary for the HSI to be greater than zero using fall VOR estimates. When over-winter VOR losses were included, the EONH in the spring declined to 21 percent, with an HSI remaining zero.

Durler/Venlo Management Unit

EONH in the Durler/Venlo unit was lower that the western SNG. EONH was reduced by eliminating the lowlands that are usually flooded in the spring from the HSI calculations. The net result was 12 percent fall EONH and 9 percent EONH in the spring. The subsequent HSI for the Durler/Venlo unit was also zero.

Table 1. Percent equivalent optimal nesting habitat and nesting HSI for three analysis areas with and without winter VOR loss on the Sheyenne National Grassland.

Analysis area	Percent EONH[1]	HSI	Percent EONH with overwinter VOR loss	HSI
Western SNG	23.8	0	19.8	0
Durler/Venlo	11.7	0	9.3	0
$\leq$1.6 km leks	25.7	0	21.1	0

[1] *EONH = equivalent optimum nesting habitat as defined in HSI model by Prose (1985).*

Area Surrounding 14 Active Leks

The area within 1.6 km of the 14 active leks had a larger EONH (26 percent) in the fall than the other analysis units. However, the nesting HSI was zero for this area as well. Four of the lek areas provided sufficient EONH for HSI's greater than zero. However HSI estimates for spring showed that only two of these leks still provided sufficient EONH for HSI's greater than zero.

DISCUSSION

Nesting HSI

Our data suggests that nesting cover limits greater prairie chicken populations on the SNG. HSI's were zero for all the analysis units we compared. Four leks had sufficient nesting cover in the surrounding 1.6 km for HSI's greater than zero based on the fall measurements. HSI for these lek areas were less than 0.2 Only two leks had HSI's greater than zero for the area within 1.6 km from leks after over winter VOR losses were considered. HSI's for these two leks were $\leq$0.1.

VOR measurements in grassland vegetation that are 2 to 3 dm are considered optimal nest cover for greater prairie chickens (Prose 1985). VOR measurements >1.5 dm provide $SI_{VOR} \geq 0.7$. Only 16 percent of the western SNG was in the VOR class >1.5 dm. In the Durler/Venlo management unit, only 7 percent of the suitable nesting area provided vegetation >1.5 dm. For areas $\leq$1.6 km of leks, only 14 percent of the area had vegetation in the >1.5 dm VOR classes. Suitable nesting cover for prairie chickens may increase during drought years because lowlands that are usually flooded are drier and usable for nesting by hens.

Most of the nesting habitat for greater prairie chickens in the SNG is the midland community type in the humocky sandhills (Manske and Barker 1981, Manske and Barker 1987). Switchgrass (*Panicum virgatum*) communities found on the toe slopes surrounding lowland meadows provide the primary prairie chicken nesting cover on the SNG (Manske and Barker 1987, Newell 1987). Although lowlands are not considered suitable for nesting in most years, the lowland/midland interface is used for nesting by prairie chickens (Newell 1987). The lack of adequate cover for nesting in upland communities was attributed to heavy livestock utilization (Newell 1987). Historically, upland communities were likely tall grass prairie (Burgess 1964), but currently have limited capacity to provide nesting cover because they are dominated by short cool season and warm season grasses such as Kentucky bluegrass and blue grama.

The HSI model (Prose 1985) assumes that optimum nesting habitat conditions exist when 80 percent of the area supports herbaceous vegetation with a VOR of 2 - 3 dm. However, lingering populations of greater prairie chickens can exist in areas with 10-15 percent permanent grassland (Hamerstrom et al. 1957, Prose 1985). Topfer et al. (1990) considers a spring population of 200 birds (100 males) as a minimum number to insure perpetuation of the population. Greater prairie chickens probably persist on the SNG because natural variation provides small limited areas with adequate nesting cover. These areas exist at the lowland/midland community interface, in lowlands during drought years, and in limited quantity surrounding some leks. Limited nesting also occurs in alfalfa on private lands (Newell 1987). Small populations, such as the greater prairie chicken on the SNG, are highly susceptible to extinction due to catastrophic natural events (Ruggiero et al. 1994).

Robustness of Analyses to Assumptions

Because the HSI in our evaluation were based on ocular estimates of VOR classes, we conducted analyses to estimate HSI for systematic errors in estimating the VOR classes. If we over estimated the VOR classes (e.g.,VOR was actually lower), then HSI would decline further. Because, the lower limit on HSI is zero, our conclusion of limited nesting habitat remained unchanged.

If we systematically underestimated VOR classes by one class (0.5 dm), HSI for the Western SNG increased to 0.1 for fall VOR estimates and remained zero for estimates of spring nesting cover. HSI in the Durler/Venlo unit remained zero for both spring and fall VOR estimates. HSI for the areas around active leks increased to 0.3 for fall VOR estimates, but declined to 0.1 for spring estimates of nesting cover. Because the area surrounding leks included lowlands that are flooded in most years, the HSI was probably lower. None-the-less, analyses that assume we underestimated nesting cover, still show that nesting habitat is limited on the SNG.

The VOR estimates we used for the 3433 ha private lands in western SNG analysis unit were made subjectively post hoc. Because, these post hoc estimates of private land VOR may have influenced the HSI, we conducted an analysis that would present the best possible HSI for this analysis unit. HSI for the western SNG was recalculated assigning all private lands with suitable vegetation types (hay and alfalfa, pastures, and CRP) for nesting, a SI_{VOR} of 1.0 (this analysis does not change the HSI for nest cover on lands managed by the SNG). The resulting HSI for nest cover increased for the western SNG analysis unit to 0.33. This HSI represents the upper limit for the western SNG analysis unit, but it is not realistic. Most of the area considered to have SI_{VOR} of 1.0 are grazed or mowed annually. Hay and alfalfa is usually cut by the third week of June, destroying existing nests and young broods unable to escape the mowers. Only the 251 ha of CRP in the analysis unit maintained its structural integrity throughout the nesting and brood rearing periods. None-the-less, this analysis still indicated that regional nesting habitat for greater prairie chickens is limited in the vicinity of the SNG.

Contributing Factors

The encroachment of woody and exotic plant species, changes in adjacent agricultural/land use changes, and livestock grazing practices are three human induced factors that directly or and indirectly influence nesting cover for prairie chickens on the SNG. Quaking aspen (*Populus tremuloides*), willow (*Salix* spp.) and Russian olive (*Elaeagnus angustifolia*) have encroached into prairie reducing nesting cover on the SNG (Kobriger et al. 1987, Jensen 1992). Leafy spurge (*Euphorbia esula*) has expanded from 7 percent to over 17 percent of the SNG since 1985 (unpubl. data, SNG). Encroachment of woody plants reduces and fragments suitable nesting, brood rearing and roosting cover (Svedarsky 1979); provides travel corridors and perch sites for predators (Burhnerkempe et al. (1984) and creates habitat more suitable for closely related sharp-tailed grouse (Prose 1987).

Agricultural development on private lands adjacent to the SNG over the past 10-15 years shows that remnant prairie habitats on private lands have been largely converted to croplands (unpubl. data, Nat. Res. Conserv. Serv., Lisbon, ND). Our analysis of the western SNG unit, showed that most of the suitable nesting habitat on private lands was Conservation Reserve Program comprising 250 ha in the analysis unit. No privately owed parcels of native prairie were identified in our analysis of the western SNG.

Grazing by livestock is the predominant use of the SNG. Livestock stocking rates have fluctuated between 50,000 and 60,000 AUMs over the past 10 - 15 years on the SNG. However, the size of livestock has increased approximately 40 percent during a comparable period (L. Potts, pers. commun., SNG, Lisbon, ND). These heavier animals require approximately 30 percent more forage (National Research Council 1984) than the standard AUM established for a 454 kg animal.

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Black-Tailed Prairie Dog Status and Future Conservation Planning

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Abstract.—The black-tailed prairie dog is one of five prairie dog species estimated to have once occupied up to 100 million ha or more in North America. The area occupied by black-tailed prairie dogs has declined to approximately 2% of its former range. Conversion of habitat to other land uses and widespread prairie dog eradication efforts combined with sylvatic plague, *Yersinia pestis*, have caused significant reductions. Although, the species itself is not in imminent jeopardy of extinction, its unique ecosystem is jeopardized by continuing fragmentation and isolation.

INTRODUCTION

The black-tailed prairie dog, *Cynomys ludovicianus* Ord, is the most widespread and abundant of five species of prairie dog in North America. Two species, the Utah prairie dog, *C. parvidens* J.A. Allen and the Mexican prairie dog, *C. mexicanus*, are currently listed as threatened and endangered, respectively, under the Endangered Species Act of 1973. The two other widespread species are the white-tailed prairie dog, *C. leucurus* Merriam and the Gunnison's prairie dog, *C. gunnisoni* Baird.

The black-tailed prairie dog is native to the short and midgrass prairies of North America. Its historic range stretches from southern Canada to northern Mexico and includes portions of Arizona, Colorado, Kansas, Montana, Nebraska, New Mexico, North Dakota, Oklahoma, South Dakota, Texas, and Wyoming (Hall and Kelson 1959). The eastern boundary of prairie dog range is approximately the western edge of the zone of tallgrass prairie, from which prairie dogs are ecologically excluded. The western boundary of this species is roughly the Rocky Mountains. Its range is contiguous with, but generally does not overlap, ranges of other prairie dog species.

With the exception of Arizona, from which it has been extirpated, the species still occurs in all the states (including Canada and Mexico) within its historic range. Yet, widespread reductions have occurred in population numbers and occupied areas throughout this broad range. Historic evidence suggests that the total area occupied by all species of prairie dogs may have declined by as much as 98% during the first half of this century (Miller et al. 1994).

METHODS

We sent letters of inquiry to state and federal conservation and land management agencies and consulted published reports. This information was augmented by telephone interviews with individuals knowledgeable about prairie dog management. The area surveyed included all states within the original range of the black-tailed prairie dog. Although responses were received from all states and agencies queried, the quality of survey information varied. Therefore, this report is a picture of prairie dogs in the mid-1980s rather than an accurate assessment of 1995 populations.

Prairie dog abundance and distribution is probably better documented at present than at any previous time due to improved mapping techniques and greater interest in prairie dogs by land management agencies. Yet, prairie dog occupied acreage can still only be grossly estimated. A primary factor contributing to this uncertainty is that much of the mapping effort is temporally distributed over a decade or more and there is no method available to assess prairie dog abundance over a broad area within a short span of time. Typically, prairie dog populations change substantially within a few years due to the threats discussed below and to climatic factors and prairie dog reproductive ecology. Another factor contributing to errors in determining prairie dog abundance is a lack of information from private and state lands.

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THREATS TO THE PRAIRIE DOG

A number of causes have been identified or proposed to account for the reductions in the acreage occupied by black-tailed and other prairie dog species. We believe that four areas of threat warrant further discussion: 1) loss of habitat due to conversion of prairie to other land uses; 2) intentional poisoning or other eradication or control efforts, primarily prompted by the livestock industry; 3) shooting for recreation or as a control effort; and 4) sylvatic plague, *Yersinia pestis*.

LOSS OF PRAIRIE

Prairie dominated by blue grama, *Bouteloua gracilis* (H.B.K.) Lag. ex Griffiths, and buffalograss, *Buchloe dactyloides* (Nutt.) Engelm., possibly due to its relatively flat topography, is among the first grassland converted to agriculture (Dinsmore 1983). As a result, Graul (1980) noted that as much as 45% of this prairie type has been lost to other land uses. Reductions in all shortgrass and midgrass prairies is expected to be similar or possibly greater in some midgrass regions where precipitation may be more suitable for agriculture. Although National Grassland acreage in the northcentral region of the Forest Service represents only about 5% of that agency's land base, it also represents the majority of the native prairie remaining in this region of North and South Dakota (Knowles and Knowles 1994).

Currently, with the exception of some areas of the northwestern portion of the black-tailed prairie dog's range, conversion of prairie to agricultural cropland has lessened. This is because much of the arable land is already in cultivation or has been converted to non-native grasses for forage. Municipal and industrial development probably account for most of the present losses to native prairies in the United States. While these losses are minor compared with those that occurred during settlement of this country, they continue to reduce habitat availability for prairie dogs and other species.

ERADICATION OR CONTROL EFFORTS

Eradication efforts have been carried out against prairie dogs on a very large scale, affecting several million ha of land (Anderson et al. 1986; Bell 1921). Clark (1979) reported that in some years prairie dogs were intentionally poisoned on more than 8 million ha in the United States. During the early 1980s, 185,600 ha of prairie dogs were eradicated on the Pine Ridge Indian Reservation in South Dakota (Hanson 1988; Sharps 1988). In 1986 and 1987, a South Dakota black-tailed prairie dog complex of 110,000 ha was destroyed, eliminating the largest remaining complex in the United States (Tschetter 1988).

Virtually every federal land management agency has been involved in this effort. The U.S. Fish and Wildlife Service used compound 1080 until its ban in 1972. In 1976, this agency approved the use of zinc phosphide as a prairie dog control agent, hoping to avoid secondary poisoning of nontarget species while maintaining its prairie dog poisoning program. It is estimated that permitting activities by both the Environmental Protection Agency and the Animal and Plant Health Inspection Service account for the annual poisoning of 80,000 ha of prairie dogs in the United States (Captive Breeding Specialist Group 1992). Much of this effort occurs on federally-owned and managed land, despite the fact that less than 5% of the United States beef weight is produced on these lands (United States General Accounting Office 1988). Most poisoning on federal land is due to private land concerns, not necessarily federal forage concerns.

The legal designation indicating the regulatory status of the black-tailed prairie dog varies among the 10 states in which it still occurs. In four states the species is designated a legal agricultural pest, with some level of either state or local mandatory controls in effect. This includes statewide legislation mandating control of prairie dogs in Wyoming. In Colorado, Kansas, and South Dakota, state legislation allows counties or townships to mandate controls on landowners. In 1995, Nebraska repealed their long-standing legislation that mandated statewide control, thereby joining the states of Montana, New Mexico, North Dakota, Oklahoma, and Texas, where control is not mandatory but assistance may be provided to landowners who believe they have a prairie dog population problem that requires control.

PRAIRIE DOG SHOOTING

Shooting of prairie dogs, either for recreation or to reduce or control their numbers, is widespread across the range of all species in the United States.

The impact this activity has on overall populations remains unclear, but preliminary monitoring results by the Bureau of Land Management (BLM) in Montana indicate that some level of shooting might impact the growth and expansion of prairie dog colonies (Reading et al. 1989). Fox and Knowles (1995) suggested that persistent unregulated shooting over a broad area of the Fort Belknap Indian Reservation in Montana might have significantly influenced prairie dog populations. However, they further concluded that it would require approximately one recreational day of shooting for every 6 ha of prairie dogs to result in such an impact. This level of shooting pressure is unlikely over the hundreds of thousands of ha of currently occupied range.

SYLVATIC PLAGUE

Prairie dogs have coexisted with a variety of predators for many centuries on the plains and have adapted means of persisting in spite of this predation. However, a more recent threat has arrived to which the prairie dog has no adaptive protection. A flea-borne bacterium, the sylvatic plague, was introduced into North America just before the turn of the century. First discovered in black-tailed prairie dogs in Texas in the 1940s (Cully 1989), small rodents such as prairie dogs apparently have no natural immunity to the plague, which now occurs virtually throughout the range of the black-tailed prairie dog.

The impacts of plague are more adverse than just the killing of many individuals. The plague persists in a colony resulting in a longer population recovery time than is common in colonies that have been poisoned (figure 1). Four years following impact, plague-killed colonies on the Rocky Mountain Arsenal National Wildlife Refuge had recovered to only 40%, while poisoned colonies had recovered to over 90% (Knowles 1986). Knowles and Knowles (1994) suggested that prairie dogs have survived the introduction of this disease simply due to their large, highly dispersed populations. Further reductions in these populations could make prairie dogs much more susceptible to local or regional extirpations due to the plague.

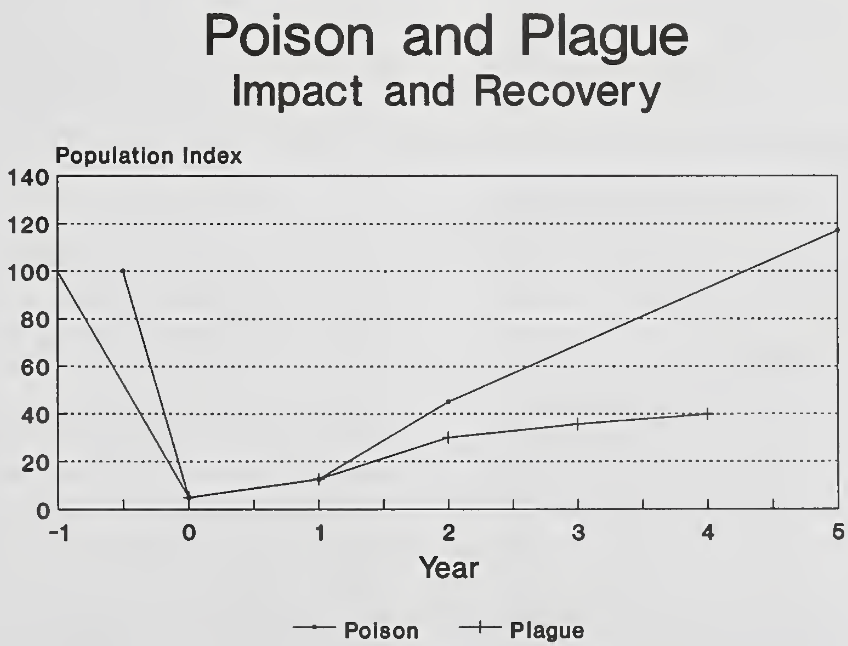

Figure 1. Comparison of prairie dog population recovery at the Rocky Mountain Arsenal National Wildlife Refuge following plague and at two colonies following control with zinc phosphide (Knowles 1986).

HISTORIC AND CURRENT STATUS

Rangewide

Seton (1929) estimated that in the early part of this century, there may have been 5 billion prairie dogs in North America. Around that time, prairie dog colonies were estimated to occupy 40 million to 100 million ha of prairie in North America, but by 1960 this area was reduced to approximately 600,000 ha (Anderson et al. 1986; Marsh 1984). These estimates result in the often-cited figure of a 98% decline in population among the five species of prairie dog. So, while the black-tailed prairie dog still occurs in all but one of the states in its historic range, significant reductions in its total colony area have taken place rangewide.

Table 1. Historic (pre-1920) and recent (post-1980) estimates of total area (ha) occupied by black-tailed prairie dogs in the United States.

State	Historic	Recent	% Change
AZ	[1]	extirpated	-100
CO	2,833,000	[1]	[1]
KS	810,000	18,845	-98
MT	595,000	35,545	-94
NE	[1]	24,415	[1]
NM[2]	4,838,460	201,220	-96
ND	85,000	8,500	-90
OK	[1]	3,850	[1]
SD	711,000	100,000	-86
TX	23,000,000	12,145	-99.9
WY	[1]	82,590	-75
United States	40,000,000 to 100,000,000	550,000	-98 to -99

[1]*Reliable data unavailable for analysis.*
[2]*Includes black-tailed and Gunnison's prairie dogs.*

PRAIRIE DOG STATUS IN EACH STATE

Current status information was solicited from state and federal agencies and from tribal authorities in all eleven states in the historic range of the black-tailed prairie dog (table 1). The following summary provides updated status and population data for those states.

Arizona

The Arizona Game and Fish Department (Duane L. Shroufe, Director, *in litt.* 1995) confirms that the black-tailed prairie dog, in the form of the Arizona subspecies *C. ludovicianus arizonensis,* is extirpated from the state. However, it still occurs nearby in Mexico and New Mexico. Arizona still supports populations of Gunnison's prairie dogs.

Colorado

On the Comanche and Pawnee National Grasslands, the Forest Service (*in litt.*) currently estimates a total of 2,455 ha of active prairie dogs, compared with 910 ha from 1978 to 1980 (Schenbeck 1982). This represents more than a doubling in area, but also represents only 0.5% of the area available on these public lands. Bent's Old Fort National Historic Site contains 325 ha of black-tailed prairie dogs (NPS, *in litt.*). Fort Carson and surrounding private lands contain approximately 1,620 ha, Pinyon Canyon less than 810 ha of prairie dogs (FWS, *in litt.*). The Rocky Mountain Arsenal NWR (FWS, *in litt.*) prairie dog population declined from 1,850 ha to 100 ha between 1988 and 1989, due to plague. Burnett (1918) estimated that three combined species of prairie dog occupied 5,665,720 ha in Colorado in the early 1900s. Based on geographic distribution of black-tailed, white-tailed, and Gunnison's prairie dogs in the state, it may be assumed that black-tailed prairie dogs accounted for approximately half this figure. There is no reliable estimate of the total area occupied by black-tailed prairie dogs statewide at this time.

Kansas

The National Park Service (*in litt.*) reports approximately 16 ha of prairie dogs at the Fort Larned National Historic Site. On the Cimarron National Grassland, the Forest Service (*in litt.*) currently estimates 440 ha of active prairie dog colonies compared with 20 ha estimated from 1978 to 1980 (Schenbeck 1982). This represents more than a twenty-fold increase on this 44,000-ha area, yet still only 1% of the total area of the Grassland. Both Lee and Henderson (1988) and Powell and Robel (1994) reported that selected counties had reductions of 84% since the beginning of the century (Lantz 1903, cited in Lee and Henderson 1988). A survey completed in 1992

(Vanderhoof et al. 1994) estimates 18,845 ha of prairie dogs in Kansas, just over 2% of the 810,000 ha estimated by Lantz (1903) some 90 years ago.

Montana

Flath and Clark (1986) estimated that black-tailed prairie dogs occupied 595,000 ha of land in Montana from 1908 to 1914. Estimated prairie dog occupied area by the early 1980s had declined to 50,600 ha (Flath and Clark 1986) and subsequent estimates show further declines in prairie dogs (40,500 ha, Campbell 1986; 35,545 ha, FaunaWest Wildlife Consultants 1995). This most recent estimate indicates a statewide reduction in occupied area of approximately 94% since the early 1900s.

Nebraska

On the Oglala National Grassland and Nebraska National Forest, the Forest Service (*in litt.*) currently estimates 105 ha of active prairie dog colonies, compared with 145 ha estimated from 1978 to 1980 (Schenbeck 1982). Current estimates represent 1.4% of land available. In 1973, prairie dog occupied area in Nebraska was estimated at 6,075 ha (Lock 1973). By 1982, this figure had increased to an estimated 32,400 ha (Frank Andelt, Nebraska Game and Parks Commission, cited in FaunaWest Wildlife Consultants 1995). By 1989, prairie dogs statewide occupied approximately 24,415 ha (Kevin Church, Nebraska Game and Parks Commission, *in litt.*). Plague and increased eradication efforts, resulting from state legislation mandating prairie dog control, have reduced this figure significantly since the 1980s, with less than 0.22% of the Nebraska landscape currently occupied by the species (FaunaWest Wildlife Consultants 1995). Historic estimates are unavailable.

New Mexico

The BLM (*in litt.*) reports that prairie dogs may be extirpated from several sites, with only 140 ha remaining on BLM land in the state. The White Sands Missile Range (Department of Army, *in litt.*) contains just over 300 ha of prairie dogs. Around 1919 the area in New Mexico occupied by prairie dogs, both Gunnison's and black-tailed (including *C. l. arizonensis*), was approximately 4,838,460 ha, but was estimated to have been reduced to 201,220 ha by 1980 (Hubbards and Schmitt 1984). This is a 96% reduction. Hubbards and Schmitt (1984) further estimated that the range of the black-tailed prairie dog in New Mexico has been reduced by one-fourth, primarily from the range of *arizonensis*.

North Dakota

Theodore Roosevelt National Park reportedly contains less than 360 ha of prairie dogs (NPS, *in litt.*), approximately 1% of the total Park land area. There are believed to be currently 2,690 ha of prairie dogs on the 660,435 ha of Custer National Forest in North and South Dakota (Forest Service, *in litt.*). This represents 0.4% prairie dog occupancy of these lands. The Forest management plan calls for an occupancy level at or around 2,225 ha. The North Dakota Game and Fish Department (*in litt.*) reports approximately 8,300 ha of prairie dogs statewide, which may be a reduction of 90% or more from historic levels. In 1992, only six complexes of over 400 ha were identified.

Oklahoma

The Department of the Army (*in litt.*) has no current estimate of prairie dog areas on Fort Sill, but report that they have declined markedly in the past 10 years. Shackford et al. (1990) reported a statewide estimate of 3,850 ha in 1967, increasing by 93% to 7,440 ha in 1989.

South Dakota

On the Buffalo Gap and Fort Pierre National Grasslands, the Forest Service (*in litt.*) estimates 3,025 ha of active prairie dog colonies and an additional 2,600 ha of colonies are subject to periodic rodenticide treatments. This compares to 17,600 ha estimated from 1978 to 1980 (Schenbeck 1982). The 500,285 ha Black Hills National Forest and Custer and Elk Mountain Ranger Districts currently support 53 ha of prairie dogs. In the early 1920s there may have been 711,000 ha of prairie dogs statewide (FaunaWest Wildlife Consultants 1995). The South Dakota Animal Damage Control office currently estimates 80,000 to 100,000 ha of active prairie dog colonies in the state; the Bureau of Indian Affairs estimates 65,000 ha of these on tribal lands (Cheyenne River Sioux Tribe, *in litt.*). These estimates suggest at least an 86% decline in prairie dog occupied area across the state. Bad-

lands and Wind Cave National Parks currently contain 1,660 and 3,085 ha of prairie dogs, respectively (NPS, *in litt.*). These numbers represent 2 and 4%, respectively, of the area available on these public lands.

Texas

There were an estimated 31,385 ha of prairie dogs in northwest Texas in 1973 (Cheatham 1973). In 1991, there were at least 12,145 ha of prairie dogs estimated in Texas (Peggy Horner, Texas Parks and Wildlife, *in litt.*). Comparing this with a statewide historic estimate of 23,000,000 ha (Merriam 1902) results in a decline of over 99% in this century.

Wyoming

On Thunder Basin National Grassland, the Forest Service (*in litt.*) currently estimates 1,500 ha of active prairie dog colonies, with an additional 4,900 ha subject to periodic rodenticide treatment. Colony area for the period 1978 to 1980 was reported to be 2,550 ha (Schenbeck 1982). These numbers represent 0.6% of this 231,500 ha public grassland area. Devil's Tower National Monument contains approximately 16 ha of black-tailed prairie dogs (NPS, *in litt.*); 3% of the area available. Black-tailed prairie dogs in Wyoming may have increased in abundance near the turn of the century as a result of sheep and cattle grazing, with an estimated 53,650 ha by 1971 (Clark 1973). However, Campbell and Clark (1981) estimated a 75% reduction in prairie dog occupied areas since 1915. Current estimates indicate between 53,000 and 82,590 ha statewide (Wyoming Game and Fish Department, cited in FaunaWest Wildlife Consultants 1995).

SUMMARY OF PRAIRIE DOG STATUS IN EACH STATE

FaunaWest Wildlife Consultants (1995) attempted to estimate the amount of land area within the range of the black-tailed prairie dog that is currently occupied by the species. They included seven Great Plains states in their analysis and concluded that the states have less than a 1% occupancy of land surface within the species' range. The states included in this assessment and the percent of prairie dog occupancy within available area are Colorado (0.35%), Kansas (0.14%), Montana (0.17%), Nebraska (0.22%), North Dakota (0.17%), South Dakota (0.80%), and Wyoming (0.60 to 0.88%).

While these individual state accounts do not represent an exhaustive rangewide status review, they unfortunately provide the best information available. Significant reductions in occupied area have and continue to occur throughout the species' range; losses in some places exceeded 95%. Although the species still occurs in all but one state in its historic range, the eastern boundary of this distribution may be receding to the west. Figures indicate that there may be more than 550,000 ha of occupied black-tailed prairie dog range remaining in the United States, which is consistent with the estimate of 600,000 ha (Marsh 1984) cited previously. Over half the known prairie dog acreage in the central and northern Great Plains occurs on private land, almost 30% is on Indian reservations, and about 6% each occurs on Forest Service and Bureau of Land Management property (figure 2, FaunaWest Wildlife Consultants 1995). Neither Park Service nor Fish and Wildlife Service lands support significant acreage of any prairie dog species.

There is a need to develop a standardized survey technique for assessing prairie dog status. Presently, two methods are commonly employed and both involve mapping of individual prairie dog colonies either by ground reconnaissance or from aerial photo interpretation. Both methods are time consuming and expensive, making it unreasonable to expect a survey of over 500,000 ha of prairie dog colonies on the Great Plains within a short time period. Prairie dog colonies represent clumped patches on a broad landscape and there already exist nonmapping techniques that might be capable of statistical sampling of this distribution (Marcum and Loftsgaarden 1980). A statistical approach to monitoring prairie dog colony acreage may be a more appropriate technique than trying to map all prairie dog colonies.

PRAIRIE DOGS AND LIVESTOCK

Efforts to eradicate the prairie dog by the livestock and agricultural industry have existed for most of this century. Merriam (1902) estimated that prairie dogs caused a 50 to 75% reduction in range productivity. Taylor and Loftfield (1924) concluded that the prairie dog is "one of the most injurious rodents of the

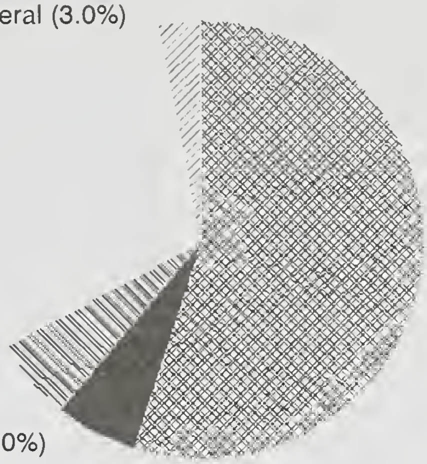

Figure 2. Distribution of black-tailed prairie dog colonies by land ownership in seven states in the northern and central Great Plains.

southwest and plains regions," and results in "the removal of vegetation in its entirety from the vicinity." Reports such as these were largely responsible for the escalating effort by range managers on the Great Plains to eradicate the prairie dog.

The conflict between the livestock industry and the prairie dog will likely not end easily or quickly, despite reports that prairie dog foraging does not significantly affect weight gain of cattle (O'Meilia et al. 1982; Hansen and Gold 1977). Others have reported the beneficial effects of prairie dogs on long-term range condition, including increased plant species diversity, richness, and overall plant production in prairie dog colonies (Archer et al. 1987; Uresk and Bjugstad 1983; Bonham and Lerwick 1976; Gold 1976). Uresk (1985) demonstrated that up to four years following prairie dog control, plant production was not increased whether the range was grazed or ungrazed by cattle.

Conversely, Hanson and Gold (1977) reported dietary overlap between cattle and prairie dogs, suggesting there may be some competition for the same species of forage plants. An estimation of true competition would be dependent on a variety of factors, including density of prairie dogs, stocking rate of cattle, ground cover, forage species present, and others (Uresk and Paulson 1988). Collins et al. (1984) reported that the annual cost of prairie dog poisoning was higher than the annual value of the forage gained by these measures. This issue requires more study, with input from both sides of the debate.

PRAIRIE DOGS AND BIODIVERSITY

The prairie dog, an integral component of the shortgrass prairie biotic community, is capable of transforming its own landscape and creating habitat alterations on a scale surpassed only by humans on the Great Plains. The ecosystem that is maintained by the prairie dog is valuable to many other species, with over 100 species of vertebrate wildlife reportedly using prairie dog colonies as habitat (Sharps and Uresk 1990; Clark et al. 1989; Reading et al. 1989). While few of these species are critically dependent on prairie dogs for all their life requisites, the increased biodiversity associated with prairie dog colonies indicates the importance of this habitat. Agnew et al. (1986) reported greater avian densities and species richness on prairie dog colonies. Also, numerous researchers have documented the preferential feeding of wild and domestic ungulates on prairie dog colonies (Coppock et al. 1983; Detling and Whicker 1987; Knowles 1986; Krueger 1986; Wydeven and Dahlgren 1985).

A number of rare and declining species are associated with prairie dogs and the habitat they provide. The black-footed ferret, *Mustela nigripes* Audubon and Bachman, 1851, is considered a true prairie dog obligate because it requires the prairie dog ecosystem for its survival. As one of the most endangered mammals in North America, this species has come to symbolize the decline in native grassland biodiversity. At least two species that are candidates for listing under the Endangered Species Act are also associated to a lesser degree with prairie dogs. The mountain plover, *Charadrius montanus* Townsend, 1837, and the swift fox, *Vulpes velox* Say, 1823, are attracted to the vegetative changes and possibly increased food availability in prairie dog colonies. The association of other species that are either declining or vulnerable indicate the problems facing this habitat.

CONSERVATION EFFORTS

Prairie dogs are managed either directly or indirectly within the survey area by at least six federal agencies, 11 state wildlife departments, state agriculture departments, departments of state lands, and numerous weed and pest districts, counties and private landowners. Prairie dog management goals and objectives vary significantly among these entities. Even management within agencies but between areas varies significantly. This variation can range from total protection of prairie dogs to a legal mandate to exterminate. All states have simultaneously classified the prairie dog as a pest and as wildlife, often with opposing management goals. Federal policy regarding prairie dogs has been inconsistent over time and across geographic regions. The legal mechanisms responsible for the decline of prairie dogs during this century are still intact. Restoration of the prairie dog ecosystem may not be possible without major changes in management policy.

At least two federal agencies have taken the initiative to begin to address the problems associated with declining prairie dog occupied areas and to involve other interested parties. The Forest Service initiated a working group comprised of various federal land and resource agencies throughout the northern states in the Great Plains, involving the Bureau of Land Management, Park Service, Bureau of Indian Affairs, and Fish and Wildlife Service. The function of this group is to encourage development of conservation assessments and strategies for the species across broad landscapes.

In January 1995, the Fish and Wildlife Service convened a meeting of federal, state, and nongovernmental entities to discuss problems facing the shortgrass prairie ecosystem, including the prairie dog as a focal species. Consensus recommendations were: 1) Fish and Wildlife Service will develop conservation strategies to keep prairie species from becoming listed under the Endangered Species Act and to recover declining species before a listing occurs; and 2) work with the Western Governor's Association to investigate ways to coordinate and communicate with all involved parties on prairie issues. The Fish and Wildlife Service recognizes that prairie dog management remains within the jurisdiction of the various state and federal land management agencies. Therefore, this agency is particularly interested in participating in cooperative agreements with other agencies so that the prairie dog may be managed as a wildlife species rather than simply controlled as a pest.

CONCLUSION

The black-tailed prairie dog does not appear to be in danger of becoming extinct in the foreseeable future, given current management. However, the additional negative impacts resulting from habitat fragmentation (Wilcox and Murphy 1985) could seriously impact the ability of some prairie dog populations to persist or become re-established. Habitat fragmentation adversely quickly affects highly specialized species (Miller et al. 1994) and the myriad of species associated with prairie dog colonies recover from habitat or population losses at different rates. This could result in a significant disruption of the ecosystem overall functioning, further delaying its recovery. Such effects are already evident for the endangered black-footed ferret. The future recovery or extinction of this species is inextricably entwined with the decisions resource managers make today regarding the conservation of the prairie dog ecosystem.

Management of the black-tailed prairie dog must give greater consideration to developing an abundance and distribution of prairie dogs that will ensure long-term population persistence of associated

species. As a minimum, we believe that broad areas of suitable grasslands should have from 1 to 3% of the area occupied by prairie dogs. Federally-owned lands should assume a greater share of this responsibility, with a goal of from 5 to 10% occupancy by prairie dogs. Maintaining this level of occupancy may allow resource managers to determine what actually constitutes a functioning prairie dog ecosystem, so attempts may be made to preserve this system into the future.

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The Role of Fire in Managing for Biological Diversity on Native Rangelands of the Northern Great Plains

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Abstract.—A strategy for using fire to manage for biological diversity on native rangelands in the Northern Great Plains incorporates an understanding of its past frequency, timing and intensity. Historically, lightning and humans were the major fire setters, and the role of fire varied both in space and time. A burning regime that includes fires at various intervals, seasons and intensities, including midsummer burns, should be reinstated. However, burning to enhance rare systems and species and to discourage exotic species is also needed. The goal is to base plans on an understanding of historic processes and ecosystem interactions, and resist techniques that rely on unexamined conventions.

INTRODUCTION

"A common thread runs through the many definitions of biological diversity: variety of life and its processes in a given area" (Salwasser 1990). A management strategy for conserving biological diversity of any natural ecosystem must focus on saving all the components, including the structure, composition (including genetic diversity), and processes that characterize these systems (Kaufmann et al. 1994). Biological diversity is more than just the identifiable parts; it also includes the symbioses and synergisms that make nature work (Salwasser 1990).

The importance of disturbances in shaping native communities has recently received more attention. Ecosystems are dynamic entities whose patterns and processes are shaped and sustained on the landscape by successional processes and by abiotic disturbances such as fire, drought, and wind. To sustain these ecosystems, processes that characterize the variability found in native ecosystems should be present and functioning, and management activities should conserve or restore historic disturbance patterns (Kaufmann et al. 1994). This paper describes a strategy for managing biological diversity of rangelands on the Northern Great Plains. The approach is based on restoring historical disturbance processes given the significantly altered landscape patterns of today. Plant nomenclature follows Great Plains Flora Association (1986) (table 1).

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SETTING

The Northern Great Plains region includes North Dakota, South Dakota and Nebraska, plus the eastern portions of Montana and Wyoming, and extends northward into Manitoba, Saskatchewan and Alberta. The climate of the region is characterized by an increase in precipitation and humidity and a decrease in periodic droughts during the summer from west to east (Risser 1990). This climate range influences not only the potential native vegetation but also the fire regime and effects. The shortgrass prairie on the Western and Southern portions of the region is the most arid type; the mixed-grass prairie occurs in the midsection of the region; and the tallgrass prairie on the Eastern edge receives the most precipitation (Risser et al. 1981).

The variation in precipitation across the region greatly influences the growth and expansion of woody plants. In the most Western portion of the region, big sagebrush occupies uplands; in the absence of fire it persists or expands (Wright and Bailey 1982). In the remainder of the shortgrass and mixed-grass portions of the region, woody plants are restricted to areas of increased elevation, such as the Black Hills, or to areas of increased moisture such as riparian zones, draws, and north-facing slopes. Escarpments, ridges, and outcrops in the Western portion support ponderosa pine and Rocky Mountain juniper (Wells 1965).

Table 1. Common and scientific names used in this report. Nomenclature follows Great Plains Flora Association (1986).

Common name	Scientific name
Graminoids	
big bluestem	*Andropogon gerardii*
smooth brome	*Bromus inermis*
cheatgrass	*Bromus tectorum*
Japanese brome	*Bromus japonicus*
buffalo grass	*Buchloe dactyloides*
threadleaf sedge	*Carex filifolia*
sand dropseed	*Sporobolus cryptandrus*
green needlegrass	*Stipa viridula*
Forbs	
leafy spurge	*Euphorbia esula*
western prairie fringed orchid	*Platanthera praeclara*
Shrubs and trees	
sagebrush	*Artemisia* spp.
dwarf sagebrush	*Artemisia cana*
big sagebrush	*Artemisia tridentata*
green ash	*Fraxinus pennsylvanica*
Rocky Mountain juniper	*Juniperus scopulorum*
Eastern red cedar	*Juniperus virginianus*
cactus	*Opuntia* spp.
ponderosa pine	*Pinus ponderosa*
plains cottonwood	*Populus deltoides*
aspen	*Populus tremuloides*
chokecherry	*Prunus virginiana*
bur oak	*Quercus macrocarpa*
willows	*Salix* spp.
snowberry	*Symphoricarpos occidentalis*

Woody draws (narrow woodlands occurring in ravines) are examples of communities in more arid portions of the region that are restricted to sites with greater soil moisture. The most common woody plants in these draws are green ash and chokecherry. Riparian zones along streams and rivers support plains cottonwood, willows, and dwarf sagebrush (Severson and Boldt 1978). These woodlands may also expand in the absence of fire, but the expansion is restricted to sites with adequate moisture and the expansion rate is slower than in the tallgrass region. Further, many deciduous species, such as chokecherry and willows, sprout vigorously following burning (Wright and Bailey 1982). Only very frequent fires (i.e., every 1 to 5 years) would favor grasses over these species.

In contrast to more arid portions of the region, mesic prairies in the Northern, Eastern and Southeastern portions of the region are characterized by precipitation amounts high enough to support the expansion of woody plants onto uplands. It is in these areas that frequent fires slow the expansion of woody plants on uplands (Bragg and Hulbert 1976). In the Northern portion of the region, aspen replaces ponderosa pine on outcrops and expands into the Canadian prairies (Wright and Bailey 1982). Eastern red cedar replaces Rocky Mountain juniper in the Southeastern part of the region where it readily expands onto uplands (Gehring and Bragg 1992). In the eastern tallgrass prairies, woody species, such as willows and bur oak, invade grasslands, and only frequent fires slow their expansion (Anderson 1990). Plains cottonwood and willow dominate floodplains in the more mesic portions of the Northern Great Plains; green ash and bur oak are common on higher terraces along major rivers (Johnson et al. 1976).

In addition to climatic factors, herbivores also influence the region's vegetation and fire regimes. However, it is difficult to distinguish the particular influence each force has on vegetation (Henderson and Statz 1995). Fire is often associated with periodic drought, and fire and grazing are sometimes interrelated. For example, recently burned grasslands often attract grazers; yet, heavily grazed areas usually resist fire until dead litter reaccumulates (Steuter et al. 1990, Vinton et al. 1993). Therefore, the influences of grazing and drought must be a part of a discussion of historical fire effects (Henderson and Statz 1995).

FIRE HISTORY

An understanding of the frequency, timing, and intensities of past fires is necessary before fire can be incorporated into a strategy to conserve prairie systems. Based on data from adjoining ponderosa pine forests, which indicated that fire frequency varied from 2 to 25 years, Wright and Bailey (1982) estimate that on level-to-rolling topography, a fire frequency of 5 to 10 years in the Northern Great Plains is reasonable. On topography more dissected with breaks and rivers, they estimate a fire frequency of 20 to 30 years. Wendtland and Dodd (1992) agree with this range, based on their examination of historical documents and fire records from the Scotts Bluff National Monument area in northwestern Nebraska. Dendrochronology data in the Devils Tower region northwest of the Black Hills reveal that before 1770 the mean interval between fires was 27 years; from 1770 to 1900 the fire return interval was 14 years (Fisher et al. 1987). Brown and Sieg (1996) report a mean fire frequency in the south-central Black Hills of 16 years for the period 1388 to 1918.

In the more mesic portions of the Northern Great Plains, the average fire return interval was shorter. Collins and Gibson (1990) estimate a frequency of every 1 to 5 years in the tallgrass portions of this region. In northcentral Nebraska, the fire return interval averaged 3.5 years between 1851 and 1900 (Bragg 1985).

Historically, the major ignition sources for prairie fires were lightning and American Indians. Lightning was, and is, an important ignition source in the Northern Great Plains. In northwestern South Dakota, lightning-set fires occur an average of 6 to 25 times per year, and most commonly occur in July and August (Higgins 1984); fewer occur in April, May, June, and September. Wendtland and Dodd (1992) note that of 10 fires described in historical documents between 1824 and 1934, and of 26 fires officially recorded between 1934 and 1969 in the Scotts Bluff National Monument area, over 70 percent occurred in July and August.

Higgins' (1986) review of 300 historical accounts written between 1673 and 1920 reveals that fires accidentally or intentionally set by American Indians were common in the Northern Great Plains. He found that although Indians set fires in nearly every month of the year, April, September and October were their peak fire-setting times. The majority of the 97 fires described were scattered, single events of short duration and small extent; only 10 fires burned longer than 1 day.

American Indians had many uses for fire. These included attracting and herding wild animals, signaling threats and warnings, improving pasturage, masking and eliminating personal signs at camps and along trails, and for pleasure, warfare and ceremonies (Higgins 1986). During their 10,000-year occupation of this region, the timing of fires set by American Indians did not mirror lightning-set fires; therefore, these Indian-set fires can be considered additive to lightning fires (Higgins 1986).

A combination of periodic droughts, high temperatures and strong winds in the region provide the components necessary for fire spread (Collins 1990). The end result of the erratic climate, flammable fuels, topographic relief and other factors, such as grazing animals, was that the role of fire was not constant in time or space (Anderson 1990).

With the arrival of non-native settlers came fire suppression policies and, in many areas, a shift in the timing of fires. Near Devils Tower, Wyoming, after 1900, the fire return interval increased to every 42 years, versus less than every 27 years previously (Fisher et al. 1987). In the south-central Black Hills, Brown and Sieg (1996) record a 104-year fire-free period in ponderosa pine stands between 1890 and 1994, and note that most of past fires occurred late in the growing season or after growth had ceased for the year. Higgins (1984) suggests that the recent extent and spread of lightning fires has been modified by cultural features such as roads; further, the fire regime has also been altered by differing patterns of grazing animals (first bison, then cattle). In contrast to the late summer ignitions that commonly burned before 1935 near Scotts Bluff, Nebraska, the 46 fires recorded since 1935 dramatically shifted to spring occurrences (Wendtland and Dodd 1992). Lengthening the interval between fires, shifting from summer to early spring burning, and/or reducing fire intensity by prescribing cooler fires may alter species composition to favor fire-intolerant species (Wendtland and Dodd 1992) such as cactus and non-sprouting woody species like sagebrush (Wright and Bailey 1982).

DEVELOPING A FIRE MANAGEMENT STRATEGY TO CONSERVE DIVERSITY

The fire strategy most likely to manage diversity on native rangelands of the Northern Great Plains is based on two premises: 1) processes that mimic, as much as possible, the variability found in native ecosystems should be present and functioning; and 2) management activities should conserve or restore historical disturbance patterns (Kaufmann et al. 1994). This management strategy should reflect the differing roles that fire historically played in the various portions of the region. However, this strategy must also address the fundamental changes that have occurred in the landscape such as drastically different landscape patterns imposed by species changes and management unit boundaries.

Wendtland and Dodd (1992) recommend a scenario that mimics the presettlement fire history. For the Scotts Bluff, Nebraska area, they infer this strategy including high intensity summer fires on a return interval of 5 to 30 years. Shifting burning programs from all spring or fall burns to include some midsummer burns should favor some species not enhanced by spring or fall burns (Howe 1994). For

example, an April fire burns early foliage critical for root production of cool-season plants, leaving late-season plants unscathed; an August fire burns the largely inactive foliage of cool-season species, while consuming foliage and reproductive stems of warm-season species (Howe 1994). However, historically, fires occurring after fuels have cured in the fall or in the early spring before green-up may have been more significant than summer fires. High fuel moisture in July and August and concurrent slow rates of spread result in a smaller area being burned by an individual fire, compared to those fires occurring when fuels are cured in the fall (Steuter 1988). Given the highly variable fire regime in the past, burns of varying intensities at differing seasons are appropriate. Further, the interval between fires should be varied to best restore fire disturbance patterns of the Northern Great Plains. The strategy should avoid a uniformity in timing of burns or in intervals between burns that artificially simplifies what was probably a more complex system (Howe 1994).

SPECIAL HABITATS AND SENSITIVE SPECIES

Reinstituting a fire regime based on historical processes that includes burning at varying intervals and in differing seasons is the first step in developing a strategy for using fire to manage biological diversity on native rangelands in this region. The second step involves assessing the direct and indirect impacts of fire on special habitats and sensitive species. Special habitats are native biological communities or ecosystems that are rare, unique, or highly productive elements of regional landscapes (Salwasser 1990). Sensitive species include those native species currently in danger of extinction or those whose population trends are negatively affected by human actions (Salwasser 1990). The burning strategy should also consider the potentially different historical fire disturbance regimes in these sensitive ecosystems, minimize potential negative influences of fire, and maximize conditions favorable to the expansion of these systems and species.

The special habitats in the Northern Great Plains (wetlands, lowlands, and riparian areas) contain high numbers of listed vulnerable species (Finch 1992, Finch and Ruggiero 1993). Although each of these habitats constitutes a relatively small percentage of the total land area, each contributes disproportionately to the diversity of native rangelands in this region (Finch and Ruggiero 1993). If sensitive communities such as these occur within a management unit, burning programs should be examined relative to their impacts on these habitats. The range in frequency, timing, and intensity of burns suitable to upland habitats may not provide optimum conditions for sustaining these distinctive systems.

Wetlands, lowlands, and riparian woodlands in this region are examples of communities that, because of higher moisture, likely burned less frequently than uplands. Riparian zones throughout the region, and woody draws in the more arid portions, tend to be green throughout most of the growing season, have higher relative humidities than adjacent grasslands, and often have running water or moist soils that slow the spread of fire into these communities. In most years, prairie fires would skip over or only burn lightly through these narrow woodlands (Severson and Boldt 1978). However, the narrow configuration and close contact of these woodlands with flammable grassland fuels suggest that historically they were exposed to a high number of grassland fires. Fire inevitably entered these woodlands, especially in dry years on hot and windy days.

Given that the species composition in woody draws includes a number of deciduous species, such as snowberry and chokecherry, that sprout following burning (Wright and Bailey 1982), and that several woody species establish best in mineral soils, fire probably functioned as a regeneration mechanism in these systems. Further, since these communities stay green longer than uplands, fires probably burned late in the growing season when there were adequate levels of cured, fine fuel. Repeated, annual fires, especially during droughts, tend to favor the growth of grasses over woody plants (Wright and Bailey 1982). Fires occurring infrequently when plants are dormant, followed by high precipitation, may enhance woody plant growth (Wright and Bailey 1982, Sieg 1991). If the goal is to regenerate woody plants in woody draws and/or to mimic historical fires, prescriptions should be set to achieve high intensities (Sieg 1996).

Rocky Mountain juniper woodlands are an example of a relatively uncommon community in the Western portion of the Northern Great Plains that rarely burned. In this region, Rocky Mountain juniper grows best on steep barren slopes (Noble 1990) where the sparse understory vegetation is rarely

adequate to sustain a fire. In areas where fine fuels are sufficient to carry a fire, the high volatile oil content of the foliage combined with Rocky Mountain juniper's inability to sprout following topkilling, results in high mortality rates (Wright and Bailey 1982).

Threatened or endangered species are examples of sensitive species whose needs cannot be ignored. Because they are the first species to drop out of ecosystems, they are considered the weakest link in the conservation of native biological diversity (Finch and Ruggiero 1993). Providing habitats in an appropriate spatial and temporal arrangement is necessary to maintain viable populations of sensitive species. Thus, vegetation management is a major tool for maintaining and restoring biodiversity, and for delisting or avoiding listing of threatened and endangered species (Kaufmann et al. 1994).

Adjusting fire management programs to meet the needs of threatened and endangered species requires an understanding of the role of fire in the long-term sustainability of the ecosystems supporting these species, and in the life history and habitat needs of individual species. For example, the western prairie fringed orchid is a federally listed threatened plant species associated with swales (low-lying often wet land) of the tallgrass prairie (U.S. Fish and Wildlife Service 1989). Although the tallgrass prairie is prone to burn every 1 to 5 years (Collins and Gibson 1990), it is unlikely that swales supporting orchids burned as often, especially during years when they were flooded. Vogl (1969) describes a "quasi-equilibrium" of a Wisconsin lowland maintained by floods during wet periods and fires during droughts. Lowlands supporting orchid populations likely burned throughout the growing season during prolonged droughts; however, fires that occur when orchids are actively growing are apt to injure or kill them. Since fall burning allows orchids to complete their life cycle, and dry conditions and lightning are inclined to occur late in the growing season, fall fires are a better choice than spring burning to sustain orchid populations and their associated habitat (Bjugstad-Porter 1993).

MANAGE INTRODUCED SPECIES

The introduction of exotic species to new environments without their associated parasites and pests may be humankind's greatest environmental manipulation (Young and Evans 1976). Many invasive exotic species have characteristics that enable them to vigorously compete with native plants and to exploit disturbed areas (Parker et al. 1993). In addition to reviewing impacts of existing non-native species and preventing the introduction of new ones (Kaufmann et al. 1994), management plans should address how to manage these species; fire is a useful tool in this arena. Problem species include those purposely planted, such as smooth brome, and a variety of species accidentally introduced, such as cheatgrass, Japanese brome, and leafy spurge (Lym 1991).

Although burning is not a panacea for discouraging introduced species, with careful planning it can be a useful tool, especially if native species are not adversely affected. Burning at a time when plants are most vulnerable is useful for suppressing undesirable species. For example, burning in mid-or late May, when smooth brome tillers are either elongating or heading, reduces tiller density of smooth brome by 50 percent when compared to unburned plots in Nebraska (Willson 1992). Burning in May also enhances production of flowering culms of some native warm-season grasses such as big bluestem (Willson 1992). However, burning is not a cure-all for reducing persistent species such as smooth brome, and the outcome is strongly dependent on other factors such as climate and precipitation patterns. Subsequent burning in Pipestone, Minnesota failed to significantly reduce smooth tiller density (Willson and Stubbendieck 1996).

In addition to killing or injuring individual exotic plants, burning can be used to make the habitat less conducive to a species expansion. Spring burning in western South Dakota killed Japanese brome seedlings for one growing season, and by reducing litter accumulations, decreased future germination rates (Whisenant and Uresk 1990). In this case, spring burning was detrimental to the production of one native species, green needlegrass; enhanced production of two others, buffalo grass and sand dropseed; and did not change the production of a fourth, threadleaf sedge (Whisenant and Uresk 1990).

A combination of burning and other management tools may be valuable in managing invasive species. For example, picloram plus 2,4-D applied in the fall followed by spring burning reduced the stem density and germination rates of leafy spurge in North Dakota more than any other treatment tested (Wolters et al. 1994). The key to success in managing invasive species is to begin treatment before expan-

sive spread occurs and to focus as much as possible on the invaded ecosystem rather than on the invader (Hobbs and Humphries 1995).

SUMMARY

A strategy for using fire to manage native biological diversity on rangelands in the Northern Great Plains should consider natural disturbance patterns. Fires historically occurred as often as every 1 to 5 years in the more mesic portions of the region, but less frequently in areas of rough topography and in lowlands. Lightning, a major ignition source in this region, caused fires most often in July and August. American Indians accidentally or intentionally set fires in nearly every month of the year; however, the greatest number were set in April, September, and October. The end result of the erratic climate, fuels, topographic relief and factors such as grazing animals, was that the role of fire was not constant in time or space.

Reinstituting a fire regime based on historical processes, including burning at varying intervals (to reflect climatic patterns) and in differing seasons, is the first step in developing a strategy for using fire to manage for biological diversity on native rangelands in this region. Including mid-summer burns, rather than concentrating all prescribed burning in the spring and fall, would better mimic natural disturbance patterns. The second step involves adjusting fire regimes to best sustain special habitats, such as wetlands and riparian zones, and sensitive species, especially threatened and endangered ones. Third, fire prescriptions should be planned so that burning does not enhance the spread of invasive species. The overall goal is to base the fire management strategy on an understanding of historic processes and ecosystem interactions, and resist techniques that rely on unexamined conventions (Howe 1994).

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